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FIRST ANNUAL REPORT ON THE ORGDP BIOLOGICAL
MONITORING AND ABATEMENT PROGRAM

OAK RIDGE NATIONAL LABORATORY

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Abstract: The monitoring and abatement program was developed for the Mitchell Branch (K-1700 stream). The document contains information about the geohydrology, water quality, toxicity monitoring and bioaccumulation studies.

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ENVIRONMENTAL SCIENCES DIVISION
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ACRONYMS

ACD	Analytical Chemistry Division
BMAP	Biological Monitoring and Abatement Program
CNF	Central Neutralization Facility
CPI	Cohort production interval
DEM	Department of Environmental Management
DO	Dissolved oxygen
DOE	Department of Energy
EFK	East Fork Poplar Creek kilometer
EFPC	East Fork Poplar Creek
EPA	U.S. Environmental Protection Agency
ESD	Environmental Sciences Division
FDA	U.S. Food and Drug Administration
GCK	Grassy Creek kilometer
GC/ECD	Gas chromatography/electron capture detector
GC/MS	Gas chromatography/mass spectrophotometry
HESA	Department of Health, Safety, and Environmental Affairs
HSRD	Health and Safety Research Division
LOEC	Lowest observed-effect level
MAF	Mean annual flow
MIK	Mitchell Branch kilometer
NOEC	No observed-effect concentration
NPDES	National Pollutant Discharge Elimination System
ORGDP	Oak Ridge Gaseous Diffusion Plant
ORNL	Oak Ridge National Laboratory
ORR	Oak Ridge Reservation
PAH	Polycyclic aromatic hydrocarbons
P/B	Production/biomass
PCB	Polychlorinated biphenyl
PCK	Poplar Creek kilometer
PGV	Preliminary guidance values
PPM	Parts per million
SD	Storm drain
TDHE	Tennessee Department of Health and Environment

TRC	Total residual chlorine
TRE	Toxicity reduction evaluation
TSCA	Toxic Substances Control Act
TSS	Total suspended solids
TVA	Tennessee Valley Authority

PREFACE

On September 11, 1986, a modified National Pollutant Discharge Elimination System permit was issued for the Oak Ridge Gaseous Diffusion Plant (ORGDP), an enriched-uranium production facility operated by Martin Marietta Energy Systems, Inc. for the U. S. Department of Energy. As required in Part III (L) of the permit, a plan for the biological monitoring of Mitchell Branch (K-1700 stream) was prepared and submitted for approval to the U. S. Environmental Protection Agency and the Tennessee Department of Health and Environment (Loar et al. 1986). The plan is referred to as the ORGDP Biological Monitoring and Abatement Program (BMAP) and describes biomonitoring activities that will be conducted over the duration of the permit. Because it was anticipated that the composition of existing effluent streams entering Mitchell Branch would be altered shortly after the modified permit was issued, sampling of the benthic invertebrate and fish communities (Task 4 of the BMAP) was initiated in August and September 1986, respectively.

This document is the first in a series of annual reports presenting the results of the ORGDP BMAP. This report describes studies that were conducted from August 1986 through December 1987. However, the actual period covered by each task or subtask varied, depending upon when the task was initiated and the turnaround time for sample analysis and data review. The studies conducted during the first year were directed toward an ecological characterization of Mitchell Branch. Although future studies will place greater emphasis on testing various hypotheses regarding the causal factors and underlying mechanisms associated with the effects documented in the initial studies, monitoring of selected parameters will continue. Any significant modifications in the parameters that are monitored or the frequency and location of this monitoring will also be addressed in the annual reports.

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This work was funded by the Department of Health, Safety, and Environmental Affairs, Oak Ridge Gaseous Diffusion Plant (ORGDP). The ORGDP and Oak Ridge National Laboratory are operated by Martin Marietta Energy Systems, Inc., under contract DE-AC05-84OR21400 with the U.S. Department of Energy.

EXECUTIVE SUMMARY

As a condition of the modified National Pollutant Discharge Elimination System (NPDES) permit issued to the Oak Ridge Gaseous Diffusion Plant (ORGDP) on September 11, 1986, a Biological and Monitoring and Abatement Program (BMAP) was developed for the receiving stream (Mitchell Branch or K-1700 stream). The objectives of the BMAP are to (1) demonstrate that the effluent limitations established for the ORGDP protect and maintain the use of Mitchell Branch for growth and propagation of fish and aquatic life, and (2) document the effects on stream biota resulting from operation of major new pollution abatement facilities, including the Central Neutralization Facility (CNF) and the Toxic Substances Control Act (TSCA) incinerator. The BMAP consists of four tasks: (1) ambient toxicity testing; (2) bioaccumulation studies; (3) biological indicator studies; and (4) ecological surveys of stream communities, including benthic macroinvertebrates and fish. This document is the first in a series of reports on the results of the ORGDP BMAP and describes studies that were conducted from August 1986 through December 1987.

Background

Mitchell Branch is a small stream which originates near the northeast boundary of the ORGDP; it flows only 1.5 km from its headwaters to its mouth at Poplar Creek kilometer 4.5. The water quality of Mitchell Branch is influenced by the geology of the drainage basin, effluents entering the stream via storm drains and the K-1407-B pond, and leachate from waste disposal sites. The water quality of lower Mitchell Branch was characterized by (1) moderate levels of dissolved solids and occasionally high levels of turbidity; (2) low nitrogen but high phosphorus loading; (3) elevated levels of most metals and some organics; and (4) elevated temperatures. The flow in lower Mitchell Branch is augmented by as much as 30% by discharges from the ORGDP. Samples were routinely collected from eight primary sites in Mitchell Branch. The toxicity monitoring and community studies included at least five of the primary sites, four of which were common among

these tasks. The uppermost site served as an undisturbed reference, and the remaining sites were selected to coincide with the ambient NPDES monitoring station or to bracket known areas or sources of ecological disturbance. Additional reference sites on area streams were also used for some tasks, depending upon the specific objective of the task.

Toxicity Testing

The toxicity of the discharges from the K-1407-B pond and storm drains (SD) 170, 180, and 190 were evaluated at approximately bimonthly intervals from October 1986 through December 1987. The toxicity of the effluents (grab samples) was determined by 7-day static-renewal tests that measured survival and growth of fathead minnow (*Pimephales promelas*) larvae, and survival and reproduction of a microcrustacean (*Ceriodaphnia*). Consistent evidence of toxicity was observed for all four discharges, although the minnows were apparently less sensitive to the K-1407-B pond effluent than to the SD discharges and less sensitive than *Ceriodaphnia* to the pond effluent. The cause of toxicity of the K-1407-B pond effluent was not identified but appeared to be related to periods of high water hardness. The toxicity of the SD discharges, on the other hand, appeared to be primarily due to high concentrations of total residual chlorine; however, dechlorination did not always reduce the toxicity of the effluents, which suggests that toxic constituents other than chlorine were sometimes present.

The ambient (instream) toxicity of six sites in Mitchell Branch was evaluated bimonthly from January through November 1987 using the same test systems as were used to evaluate the point source discharges. Water from the upstream reference site at Mitchell Branch kilometer (MIK) 1.43 adversely affected fathead minnows but not *Ceriodaphnia*. Because there are no obvious sources of toxicants at this site, a bacterial or fungal fish pathogen may be involved; this hypothesis is currently being tested. Water from MIK 0.71 downstream to MIK 0.12 showed definite evidence of toxicity to both test organisms, thus indicating the strong influence of the K-1407-B pond effluent and the SD170, SD180, and SD190 discharges. Due to the considerable variability in survival of the test animals and to a lack of consistent evidence for

chronic toxicity, it is hypothesized that the ambient toxicity pattern exhibited in Mitchell Branch may be determined by episodic releases of one or more toxicants. Information to date suggests that chlorine may dominate the pattern of ambient toxicity observed in lower Mitchell Branch.

Bioaccumulation Studies

Efforts were initiated to (1) identify any substances being discharged into Mitchell Branch that accumulate in Mitchell Branch biota in excess of those observed in biota from nearby, uncontaminated streams; (2) evaluate the extent and significance of contamination by those substances in Mitchell Branch and downstream aquatic systems; (3) assist in locating sources of contaminants that accumulate to unacceptable levels; and (4) distinguish the relative importance of ongoing discharges versus historical or residual contamination in determining contaminant levels in biota. Because of the low abundance of adult game or food fish populations in Mitchell Branch, caged clams (*Corbicula fluminea*) were used for most analyses; where possible however, redbreast sunfish (*Lepomis auritus*) and bluegill sunfish (*L. macrochirus*) were also used.

Contaminants that were clearly elevated in the biota of Mitchell Branch included mercury and polychlorinated biphenyls (PCBs). The estimated maximum concentration of mercury in redbreast sunfish of Mitchell Branch (0.3 to 0.5 ppm, wet weight) was however, below the U.S. Food and Drug Administration (FDA) action limit of 1.0 ppm. Similar concentrations of mercury were found in sunfish from nearby Poplar Creek and lower East Fork Poplar Creek. It was not possible to determine from the available data, however, whether Mitchell Branch was a source of elevated mercury concentrations in its fish.

PCB monitoring data showed conclusively that Mitchell Branch was a source of PCB contamination in 1987 to biota inhabiting the stream, and may have been and could continue to be a significant source of PCB contamination in lower Poplar Creek. Levels of PCBs in caged clams in Mitchell Branch (2.5 to 3.9 ppm, wet weight) exceeded the FDA limit of 2 ppm. Concentrations of PCBs in sunfish in Poplar Creek

(0.17 to 0.22 ppm) were below the FDA limit but high enough to suggest that, in other species of fish that have a higher propensity to accumulate PCBs, concentrations in some individuals may exceed this limit. The K-1407-B pond was found to contain PCBs in early 1987. The source of PCBs to the pond was subsequently identified and eliminated, and aqueous phase concentrations have since remained below detection limits; however, the detection limit of these analyses was well above biologically significant concentrations. Whether Mitchell Branch is still a major source of PCBs to Poplar Creek remains to be determined. [Results of more recent studies indicate that (1) the concentration of PCBs in caged clams in Mitchell Branch has decreased substantially since 1987 and (2) no striking increase in the concentration of PCBs in bluegill from lower Poplar Creek was associated with the 1987 discharge of PCBs to Mitchell Branch.]

Biological Indicator Studies

Both quantitative studies and several attempts to collect fish showed that insufficient numbers and sizes of fish occurred in Mitchell Branch to accomplish the biological indicator task during this reporting period. Therefore, studies will be initiated in the second year which will involve the placement of caged and uncaged tagged fish into the lower reaches of Mitchell Branch. Several fish will then be collected periodically and a suite of biological indicators will be measured.

Instream Ecological Monitoring

Benthic macroinvertebrate communities were sampled monthly from August 1986 through July 1987 at six sites in Mitchell Branch; fish communities were sampled quarterly from September 1986 through October 1987 at five sites in Mitchell Branch and one site in Grassy Creek, a nearby reference stream. Results of these studies supported the findings of the ambient toxicity studies. Maximum impact to both the benthic and fish communities was observed at MIK 0.71, a site located just downstream of SD170. Some minor recovery of these communities was evident at those sites downstream of MIK 0.71. Maximum density, taxonomic richness, and diversity of benthic invertebrates were found in

the reference site (MIK 1.43), while some impact was evident at both MIK 0.86 and MIK 0.78. The fish community on the other hand, exhibited maximum density and biomass at MIK 0.78. Fish abundance and biomass at this site were also greater than in Grassy Creek. MIK 0.78 appears to serve as a refuge for fish from toxic discharges to downstream reaches of Mitchell Branch, which could therefore result in unnaturally high fish densities and biomass. Siltation is possibly the primary source of impact in the upper reaches of Mitchell Branch below MIK 1.43. One of the more evident stressors to fish and benthos downstream of SD170 is chlorine, but other potential stresses downstream of this storm drain were also identified including elevated levels of metals, siltation, and high temperatures.

Future Studies

Results of studies conducted during the first year of the ORGDP BMAP will be used to guide future monitoring efforts. Sampling sites and frequencies will remain the same for the effluent and ambient toxicity studies, and the benthic macroinvertebrate and fish studies. Caged clams and small resident fish will continue to be used in the bioaccumulation studies, while caged and introduced fish will be used in the biological indicator study and possibly the bioaccumulation studies as test organisms.

Although the detailed characterization of Mitchell Branch will continue in the second year, increasing emphasis in the second and subsequent years will be placed on the development and testing of hypotheses on the factors and mechanisms contributing to the adverse conditions observed during the first year. For example, studies will be initiated to determine the contributions of several storm drains to the total flow of Mitchell Branch in order to obtain a better estimate of their importance to patterns of instream toxicity. Efforts will be initiated to develop a better understanding of the role of Mitchell Branch as a source of PCBs to lower Poplar Creek and to identify the source of PCBs to Mitchell Branch. Ultimately, the rate of recovery of the biotic communities, the elimination of toxicity in the middle

reaches of Mitchell Branch, and the reduction of contaminant residues in biota of Mitchell Branch will all depend upon accurate identification of the causal factor(s).

1. INTRODUCTION

On September 11, 1986, a modified National Pollutant Discharge Elimination System (NPDES) permit was issued for the Oak Ridge Gaseous Diffusion Plant (ORGDP), an enriched-uranium production facility operated by Martin Marietta Energy Systems, Inc. for the U.S. Department of Energy (DOE). As specified in Part III (L) of the permit, a plan for the biological monitoring of the receiving stream (K-1700 stream or Mitchell Branch) was submitted for approval to the U.S. Environmental Protection Agency (EPA) and the Tennessee Department of Health and Environment in December 1986. Because it was anticipated that the chemical composition of several effluents could be altered shortly after the permit modifications were issued, the ORGDP Biological Monitoring and Abatement Program (BMAP) was implemented in August 1986, before formal approval of the plan was received from the regulatory agencies.

The BMAP for Mitchell Branch was developed to meet two major objectives. First, studies were designed to provide sufficient data to demonstrate that the interim effluent limits established for the ORGDP protect and maintain the use of Mitchell Branch for growth and propagation of fish and aquatic life. A second major objective is to document the effects on stream biota resulting from construction and operation of major new pollution abatement facilities, including the Central Neutralization Facility (CNF) and the Toxic Substances Control Act (TSCA) incinerator. The ecological effects of remedial actions (e.g., closure of the K-1407-B and K-1407-C holding ponds) can also be evaluated by this monitoring program.

The effluents discharged to Mitchell Branch are complex and include various trace metals, organic chemicals, neutral salts, and radionuclides (Sect. 2.2). Moreover, the composition of these effluent streams will change as various pollution abatement measures are implemented over the next several years. Although contaminant inputs to the stream originate primarily as point-sources from existing plant operations, area or non-point sources such as the classified burial grounds and the K-1407-C holding pond cannot be eliminated as potential sources of contaminants. A multi-tiered, integrated approach to biological monitoring was developed to address this level of environmental complexity. The BMAP for Mitchell Branch consists of

three major tasks: (1) effluent and ambient toxicity testing; (2) bioaccumulation studies; and (3) ecological surveys of stream communities (e.g., benthic macroinvertebrates and fish). Because of the low abundance of fish in Mitchell Branch, a fourth task on biological indicators of stress will be conducted using a hatchery species that can be introduced into the stream in enclosures. Because of the prolonged drought and its potential effects on biota, especially in the reference streams where there is no flow augmentation, implementation of this task has been delayed.

This report describes the results of studies conducted during the first year (August 1986-July 1987), although some data collected through December 1987 are also presented.

2. DESCRIPTION OF STUDY AREA

Mitchell Branch is a small, second-order stream located near the northeast boundary of the ORGDP (Fig. 2-1). With a drainage area of 1.78 km², it is similar in size to upper Grassy Creek [2.59 km² at Grassy Creek kilometer (GCK) 2.4], a relatively undisturbed reference stream located approximately 2 km southeast of the ORGDP (Fig. 2-2). Mitchell Branch flows only 1.5 km from its headwaters to its confluence with Poplar Creek about 150 m downstream of the Blair Road bridge. The confluence of the two streams is about 1.5 km below the mouth of East Fork Poplar Creek (EFPC) and 7 km above the confluence of Poplar Creek with the Clinch River.

2.1 GEOHYDROLOGY

Mitchell Branch originates near the base of a small knoll southwest of McKinney Ridge. The knoll is underlain by the Conasauga Group which consists of a calcareous shale interbedded with thinner layers of limestone and siltstone (DOE 1979). Basins like Mitchell Branch that are underlain by shale and sandstone have a smaller low-flow discharge and greater range in flow than basins underlain by carbonate rocks (e.g., Knox Dolomite) (McMaster 1967). Periods of zero discharge have been observed in portions of Mitchell Branch just upstream of the BMAP sampling site at Mitchell Branch Kilometer (MIK) 0.86 [J. G. Smith, Health and Safety Research Division (HSRD), Oak Ridge National Laboratory (ORNL), personal observation]. Periods of zero discharge are also characteristic of the upper reaches of Melton Branch near ORNL (Lowery et al. 1987), a small basin that is also underlain predominantly by shale and sandstone (McMaster 1967, Table 10).

While no flow was observed in portions of the upper reaches of Mitchell Branch, discharges from the ORGDP augment the flow below MIK 0.78. Once-through cooling water and process water account for about 21% and 10%, respectively, of the stream flow at NPDES monitoring station K-1700 on lower Mitchell Branch at MIK 0.12 (Kasten 1986). Surface runoff probably accounts for most of the remaining flow (69%) (Kasten 1986), although groundwater also contributes to surface flow

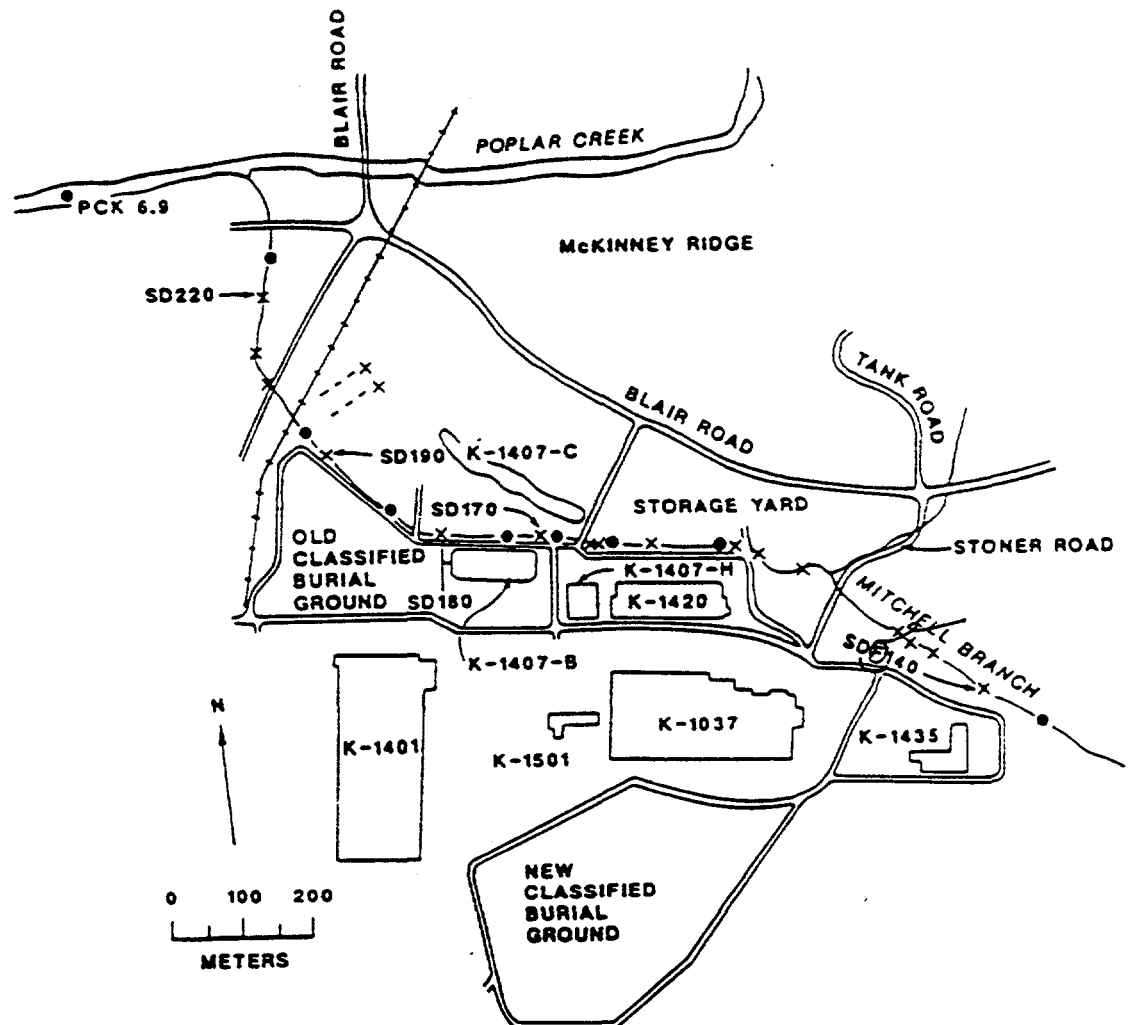


Fig. 2-1. Map of Mitchell Branch and a portion of the Oak Ridge Gaseous Diffusion Plant showing the locations of the biological monitoring sites (•) in relation to the outfalls of the storm drains (x).

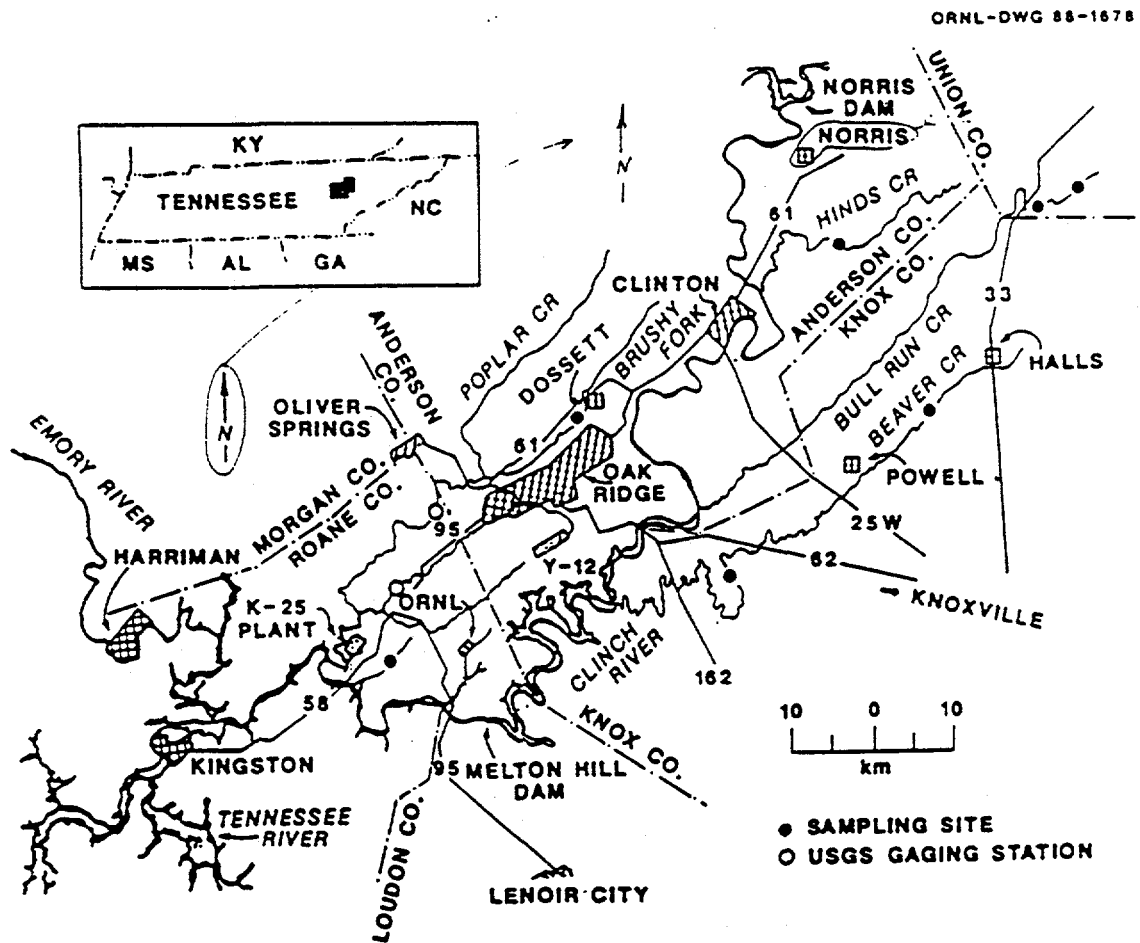


Fig. 2-2. Map of the Oak Ridge area showing locations of the reference (control) sites.

(Scheib 1987, Table 7). Based on these estimates of average water use, approximately 31% of the flow in lower Mitchell Branch can be attributed to discharges from the ORGDP. However, in years of below-normal precipitation, such as occurred in 1985-1987, and minimal runoff, nearly 100% of the flow in the stream can be plant effluent. The benefit to biota derived from increasing the minimum flow in the stream (i.e., stream bed dewatering and habitat loss are reduced) can be offset by the adverse impacts of minimal or no dilution of these effluent.

Another characteristic of the hydrology of Mitchell Branch is the lower temporal variability in discharge volume compared to streams without flow augmentation (Table 2-1). The variability in discharge was higher than EFPC at East Fork Kilometer (EFK) 5.3 where flow augmentation is approximately 50% in dry years. The effect of flow augmentation on temporal variability may be beneficial. For example, Horowitz (1978) found that greater numbers of fish species occurred in streams where flow was more constant.

The mean annual flow (MAF) in Mitchell Branch decreased by approximately 25% each year from 1985 to 1987 (Table 2-1). Placing the ORGDP on standby status during this period reduced the discharges of small, once-through cooling systems to the stream by almost 50% (Kasten 1986, Fig. 12). Because these systems account for less than 25% of the flow in Mitchell Branch, they probably contributed less to the reduction in stream discharge than the below-normal precipitation that occurred each year since 1985. Generally, streams with and without flow augmentation decreased in flow from 1985 to 1987 (Table 2-1). Even in 1985, which was used as a baseline for this comparison, the MAFs in Poplar Creek and EFPC were only 62% and 79%, respectively, of the historical MAF (period of record: 1960-1985) (Lowery et al. 1987).

Although precipitation was below normal in each of the last three years, the Mitchell Branch hydrograph in 1985 differed substantially from those in 1986 and 1987 (Fig. 2-3). Flows were generally less variable in 1985 (Table 2-1), and there was no prolonged low-flow period which was a dominant feature of the 1986 and 1987 hydrographs. During a five-month period from summer through early fall, mean monthly flows were below 450 and 350 L/min in 1986 and 1987, respectively; for two

Table 2-1. Comparison of mean annual discharge (L/s) and variability in discharge for local streams with and without significant flow augmentation. The mean and coefficient of variation (CV) were computed from mean monthly values taken from monthly NPDES reports (Mitchell Branch at MIK 0.12 and East Fork Poplar Creek at EFK 23.7), Lowery et al. (1986, 1987), and USGS provisional discharge data (L.D. Voorhees, ORNL, Environmental Sciences Division (ESD), unpublished data)

	1985		1986		1987		% change			
	\bar{X}	CV	\bar{X}	CV	\bar{X}	CV	1985 to 1986	1986 to 1987	1985 to 1987	
<u>With flow augmentation</u>										
Mitchell Branch	16.7	50.9	12.5	81.6	9.6	70.8	-25.2	-23.2	-42.5	
East Fork Poplar Creek										
EFK 23.7 ^a	412	14.6	412	6.6	357	22.1	0.0	-13.4	-13.4	
EFK 5.3	1114	33.8	1011	52.1	998	50.6	-9.2	-1.3	-10.4	
<u>Without flow augmentation</u>										
Brushy Fork ^b	370	63.2	351	115.3	283	109.7	-5.1	-19.4	-23.5	
Upper Melton Branch	c	-	10.5	137.1	10.2	148.4	-	-2.9	-	
Bear Creek	c	-	107	110.6	111	110.6	-	+3.7	-	

^aOutfall of New Hope Pond at the Y-12 Plant.

^bEstimated as $Q_{BF} = A_{BF}/A_{PC} * Q_{PC}$, where A_{BF} and A_{PC} are the areas of the Brushy Fork and Poplar Creekwatersheds and Q_{PC} is the stream flow at the USGS gaging station on Poplar Creek at PCK 22.2 (Fig. 2-2).

^cNew USGS station; no records prior to April 1, 1985, for Melton Branch and or before March 1, 1985, for Bear Creek.

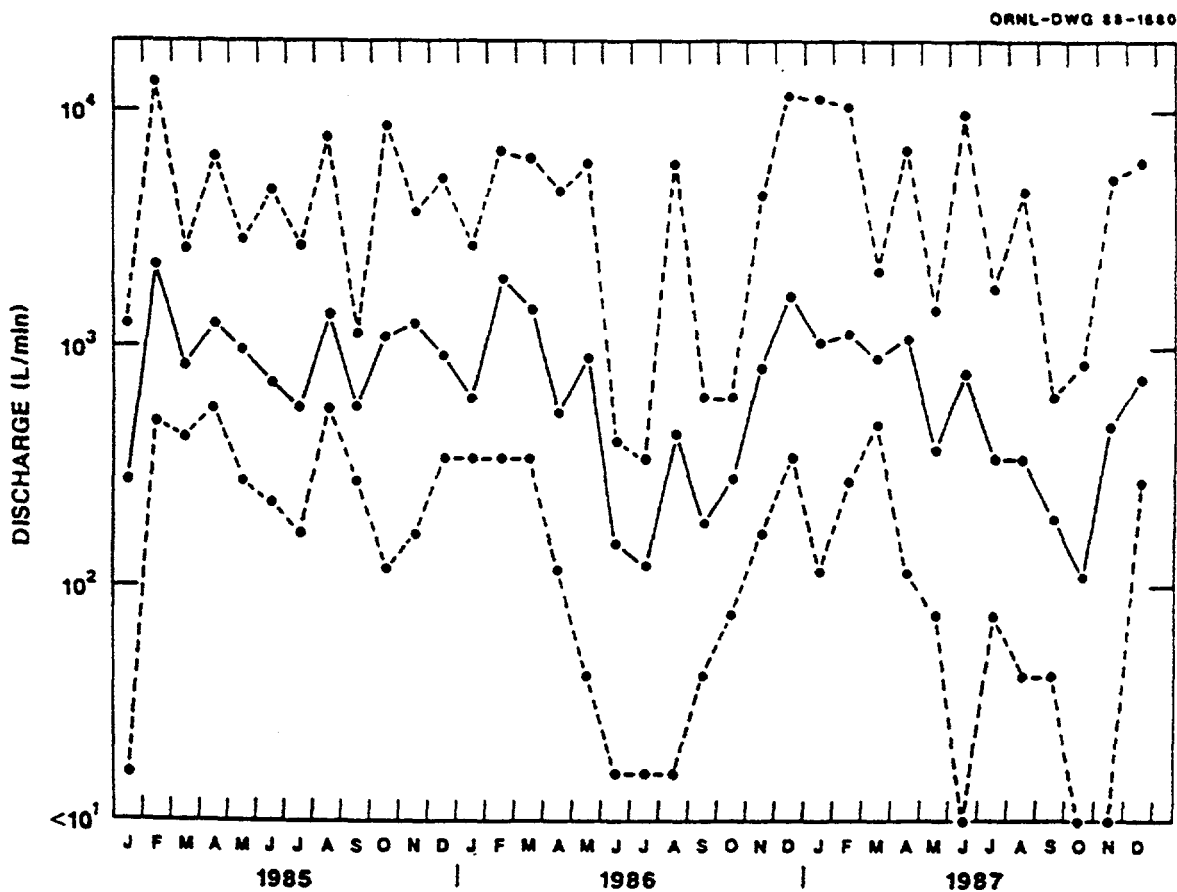


Fig. 2-3. Mean monthly stream flow in Mitchell Branch at MIK 0.12 (NPDES monitoring station K-1700) below the Oak Ridge Gaseous Diffusion Plant. Flow measurements were obtained daily, except on weekends and holidays. Maximum and minimum values for each month are designated by the dotted lines. Source: NPDES Monthly Reports

consecutive months in both years, flows averaged less than 150 L/min. In comparison, mean monthly flows in 1985 never fell below 500 L/min and were never less than 700 L/min for more than two consecutive months. Also, the minimum daily flow in 1985 was almost an order of magnitude higher than the minimum flows in 1986 and 1987.

The number of days of zero discharge in upper Melton Branch also provides a relative measure of ecologically meaningful differences in the hydrographs of the past three years. For example, no flow was recorded for 104 d in 1986 and 172 d in 1987 (Table 2-2). More importantly, the number of consecutive days with zero discharge in 1987 (155 d) exceeded the number in 1986 and 1985 by factors of three and 25, respectively. These data imply that any adverse ecological effects of reduced stream flow were also greater in 1987 due to below-normal precipitation that year and the previous year. Thus, the first year of biological monitoring was conducted over extreme, and possibly worse-case, ambient conditions.

2.2 WATER QUALITY

Water quality of Mitchell Branch is influenced not only by the geology of the drainage basin (Sect. 2.1) but also by (1) effluents that enter the stream via the K-1407-B pond and storm drains, (2) leachate from waste disposal sites (i.e., area-source discharges), and (3) remedial action projects. A general description of the sources of the effluents discharged to the stream precedes the water quality characterization, which is based on the NPDES monitoring at station K-1700 (Sect. 2.2.2), a storm drain survey conducted in 1987 (Sect. 2.2.3), and BMAP-related measurements of water temperatures (Sect. 2.2.4).

2.2.1 Description of ORGDP Discharges to Mitchell Branch

Point-source discharges to Mitchell Branch from current ORGDP operations generally fall into one of two categories: process water or storm drain effluents. Most of the process water is discharged to the stream via the K-1407-B retention basin, a 0.54-ha holding pond constructed primarily for the settling of solids and pH control. The

Table 2-2. Number of days of zero discharge (number of consecutive days in parentheses) in upper Melton Branch at USGS gaging station 03537100 near ORNL. Source: Lowery et al. (1986, 1987) and USGS provisional data (L. D. Voorhees, ORNL, unpublished data)

	June	July	August	September	October	November	December	Total
1985	4 (2)	8 (6)	0	2 (2)	0	0	0	14 (6)
1986	10 (10)	15 (15)	31 (31)	27 (24)	21 (12)	0	0	104 (47)
1987	11 (8)	21 (19)	31 (31)	30 (30)	31 (31)	30 (30)	18 (14)	172 (155)

basin receives (1) uranium compounds and acidic/caustic wastes discharged from the decontamination and recovery facility (Building K-1420), (2) neutralized wastes from the metals preparation facility (K-1401), (3) caustic wastes from the steam plant water treatment process, and (4) coal yard runoff. The overflow discharge from the basin joins with storm drain 180 (SD180) and enters Mitchell Branch at MIK 0.63. Prior to August 1987, some coal yard runoff and boiler blowdown was discharged directly to the stream via SD170 just upstream of the K-1407-B pond (Fig. 2-1).

Eighteen storm drains enter Mitchell Branch (Fig. 2-1). Although some of these drains contribute only suspended particulate matter to the stream during rainfall events (Table 2-3), others may discharge groundwater, once-through cooling water, or floor drain wastewater in addition to runoff from roofs and parking lots. In the NPDES permit, storm drains are classified according to their source and potential for contamination. Nine storm drains at the ORGDP are classified as category III outfalls (those that may receive untreated process wastewaters) and seven of these nine drains discharge to Mitchell Branch.

Leachate from waste disposal sites (i.e., area-source discharges) may also enter the stream. The old classified burial ground, a 1.50-ha site located 120 m west of the K-1407-B pond, was created by filling in a large swampy area that drained into Mitchell Branch. This disposal site contains both radioactive and nonradioactive wastes. Classified wastes are presently disposed of at a site just south of the K-1501 steam plant (Fig. 2-1). An ephemeral stream drains the site, which is located within the Mitchell Branch watershed. The K-1407-C retention basin has an area of 0.80 ha and is located 120 m northeast of the K-1407-B pond. It was constructed in 1973 and received dredged material from other holding ponds, including K-1407-B. The basin contains dewatered sludge contaminated with low levels of radioactivity, primarily uranium. Although the basin has no surface effluent, a groundwater plume extending from the pond toward the stream has been detected (Ashwood et al. 1986).

Table 2-3. Categories and sources of the 18 storm drains that enter Mitchell Branch above NPDES monitoring station K-1700 (MIK 0.12). The storm drains were sampled between March and July 1987. NS - Not sampled due to lack of flow. Source: Scheib (1987, Tables 3, 7, and 8)

Category ^a	Storm Drain																	
	140	142	144	148	149	150	152	160	162	168	170	180	190	192	194	200	210	220
	III	III	III	II	II	II	II	II	III	II	III	III	III	I	I	II	II	I
Source																		
Groundwater	X										X	X						
Roof drains	X	X					X				X	X	X			X	X	X
Area runoff																		
General	X	X	X	X	X	X	X	X	X		X	X	X	X	X		X	X
Oil storage area																		
Sludge fixation area										X								
Garage area					X							X						
Floor drains (K-1401)												X	X					
Once-through cooling water												X	X			X		
Cooling tower blowdown											X	X						
Autoclave condensate											X					X		
No. samples collected	6	5	4	NS	NS	5	NS	1	2	NS	4	6	6	NS	2	2	6	NS

^a Storm drains were categorized in the NPDES permit on the basis of effluent source: Category I includes drains that were installed to control runoff; Category II includes drains that receive (1) runoff from parking lots and roofs and (2) cooling water discharges, including once-through cooling water, cooling tower blowdown, and condensate; and Category III includes building drains that may be contaminated by pollutants because of inflow-infiltration, cross-connections, or improper disposal of chemicals (DEM 1986; Scheib 1987).

Several significant remedial actions are planned that will alter the existing water quality of Mitchell Branch. The new CNF (Building 1407-H), which is scheduled to be on-line in 1988, will treat effluents from the decontamination and recovery facility, the metals preparation facility, and the TSCA incinerator (Building K-1435). When operational, the TSCA incinerator will be used to dispose of polychlorinated biphenyls (PCBs) and other hazardous wastes. A second major waste stream to the CNF consists of coal yard runoff and boiler blowdown from the K-1501 steam plant. Modifications will be made to the CNF in 1988 that will permit replacement of the K-1407-B pond, which is scheduled for closure under the Resource Conservation and Recovery Act in November 1988.

2.2.2 NPDES Monitoring of Mitchell Branch

The following characterization of water quality is based on the routine monitoring of 24 parameters at NPDES station K-1700. This site is located on lower Mitchell Branch (MIK 0.12) downstream of all point- and most area-source discharges. With the exception of total halomethanes, all the parameters are monitored at least twice weekly (Table 2-4).

Water quality in Mitchell Branch in 1985 and 1986 was characterized by (1) moderate levels of dissolved solids and occasionally high levels of turbidity, (2) relatively low levels of nutrient enrichment, (3) elevated levels of most metals and some organics, and (4) high temperatures, as discussed in Sect. 2.2.4. Total dissolved solids averaged 340 mg/L from January through July 1985, and 553 mg/L for the rest of the year (W. J. Scheib, unpublished data from 1985 NPDES monthly reports). Average annual levels also exceeded 500 mg/L in both 1986 and 1987 (Tables 2-4 and 2-5, respectively). High turbidity was usually associated with high stream flows, although runoff from construction sites adjacent to Mitchell Branch contributed to the periodically high suspended loads in the stream.

Mitchell Branch is also characterized by low nitrogen and high phosphorus loading. Even though concentrations of nitrate nitrogen were higher than background (~0.1 mg/L; Boyle et al. 1982, Table 3.16), they

Table 2-4. Median and range of concentrations of the 24 NPDES parameters measured in water from lower Mitchell Branch (MIK 0.12) in 1985 and 1986. Values were computed from the average, maximum, and minimum monthly values

Parameter	Type of sample ^a	Sampling frequency ^b	1985		1986		
			Median	Range	Median	Range	
Organics, µg/L							
Halomethanes, total	2	3	<10	all<10	<10	5-9	
Methylene chloride	2	1	<3	<3-13	<5	<5-15	
Tetrachloroethene	2	1	<5	<4-26	<5	<5-13	
1,1,1-trichloroethane	2	1	5	<4-20	<5	<5-32	
Trichloroethene	2	1	40	<2-150	60	<5-130	
Metals, µg/L							
Aluminum ^c	1	1	0.40	0.10-12.4	0.38	0.02-11.0	
Beryllium	1	1	<1	all<1	<1	<0.3-<3	
Cadmium	1	1	<2	<2-13	<2	<2-4	
Chromium	1	1	<10	<10-50	10	<10-30	
Lead	1	1	<5	<4-20	6	<4-24	
Mercury	1	1	<1	<0.2-<1	<0.2	<0.2-0.7	
Selenium	1	1	<5	<2-60	<5	<5-8	
Silver	1	1	<10	all<10	<10	all<10	
Zinc	1	1	50	<20-560	30	4-100	
Conventional Parameters, mg/L							
Chemical oxygen demand ^d	1	2	11	<5-59	8	3-51	
Dissolved solids ^d	1	1	376	133-981	604	175-2,245	
Flow, L/min ^d	2	4	943	15-13,319	562	16-11,830	
Fluoride	1	1	0.35	0.15-1.10	0.40	<0.10-1.5	
Nitrate nitrogen ^d	1	1	0.92	0.27-4.85	0.40	<0.11-5.7	
Oil and grease	2	1	<2	<2-5	2	<2-11	
pH	3	4	7.4	6.1-8.9	7.5	6.1-9.0	
Suspended solids	1	2	16	1-62	9	<1-47	
Temperature, °C	2	2	15.5	5.5-25.0	17.5	4.5-24.5	
Turbidity, NTU ^{d,e}	2	2	12	2-260	10	2-200	

^a1-24-h composite; 2-grab; 3-continuous.

^b1-two times per week; 2-four times per week; 3-four times per year; 4 - daily.

^cmg/L.

^dMedian was greater than $\pm 10\%$ of mean; mean values \pm standard deviation for these parameters in 1985 and 1986, respectively, were: chemical oxygen demand (10 ± 3 and 9 ± 2 mg/L); dissolved solids (429 ± 128 and 627 ± 159 mg/L); flow (997 ± 506 and 753 ± 617 L/min); nitrate nitrogen (0.98 ± 0.33 and 0.50 ± 0.28 mg/L); and turbidity (14 ± 7 and 12 ± 9 NTU).

^eNTU = Nephelometric Turbidity Unit.

averaged less than 1 mg/L and never exceeded 8 mg/L at MIK 0.12 (Tables 2-4 and 2-5). Phosphorus is not monitored in Mitchell Branch but was measured in the storm drain survey (Sect. 2.2.3) and in the effluent from the K-1407-B pond (Sect. 3.1.3.1). Levels were at or near the detection limit by inductively-coupled plasma analysis of 0.2 mg/L in 12 of the 13 storm drains sampled. In SD170, however, which receives cooling tower blowdown (Table 2-3), the average and maximum concentrations were 0.8 and 4.1 mg/L, respectively. The median phosphorus concentration in the K-1407-B pond effluent was 1.51 mg/L, although substantially higher concentrations occur (Sect. 3.1.3.1).

Seven of the nine metals that were measured were present in Mitchell Branch at concentrations exceeding levels (Tables 2-4 and 2-5) typical of small, relatively undisturbed streams on the Oak Ridge Reservation (Boyle et al. 1982, Tables 3.16 and 4.16). In the more than 300 water samples that were analyzed from lower Mitchell Branch between 1985 and 1987, beryllium and silver were always below the detection limit of 1 and 10 $\mu\text{g/L}$, respectively. With the exception of aluminum, lead, and zinc, all metals had a median concentration that was at or below the detection limit in each of the past three years. Four organics and total halomethanes are monitored in Mitchell Branch, and all were above detection limits at least some of the time. Except for the increase in total dissolved solids that occurred in July 1985, the water quality of Mitchell Branch in 1985 and 1986 was similar to that in 1987.

2.2.3 Storm Drain Survey

A comprehensive water quality survey of the ORGDP storm drain system was conducted from March through July 1987 (Scheib 1987). Because rainfall during this period was below normal, several drains were dry. Data were obtained on 13 of the 18 storm drains that discharge to Mitchell Branch. Based on discharge volume and sources, the most significant discharges are SD170, SD180, and SD190.

A comparison of the NPDES water quality parameters measured in Mitchell Branch at K-1700 (MIK 0.12) in 1987 and in SD170, SD180, and SD190 is shown in Table 2-5. Compared to the other drains and to

Mitchell Branch, chromium was higher in SD170 and mercury was higher in SD180 and SD190. In general, however, concentrations of the NPDES parameters in the storm drains were similar to those in the stream.

Some storm drains that flow into Mitchell Branch contained 10 to 100 $\mu\text{g/L}$ of organic solvents (chlorinated ethenes and ethanes), halogenated methanes (likely to be present in chlorinated process water), phthalates, freons, hydrocarbons, and other industrial solvents; a number of unidentified compounds were also detected. The metals cleaning facility (K-1401) uses trichloroethane (DOE 1979) which enters Mitchell Branch via floor drains to SD180 and SD190 (Table 2-3), although 1,1,1-trichloroethane was also present in SD170 (Table 2-5). All three outfalls contained halogenated methanes, principally chloroform and dichlorofluoromethane. Except for naphthalene, which was present at the detection limit of 10 $\mu\text{g/L}$ in SD180, no polycyclic aromatic hydrocarbons (PAHs) or PCBs were detected. High levels of aliphatic and aromatic hydrocarbons (1100 and 220 $\mu\text{g/L}$, respectively), total organic carbon (115 mg/L), and oil and grease (315 mg/L) were found in SD140 following a fuel oil spill that was first noted on March 13, 1987 (W. J. Scheib, HSEA/ORGDP, personal communication). Substituted diphenyl phosphate, an herbicide (Prometon), and several groups of unknown organic compounds (total of 193 $\mu\text{g/L}$) were detected in SD180, which drains the classified burial ground and garage area.

Inputs of contaminated groundwater to several storm drains and to Mitchell Branch is indicated by the presence of tetrachloroethene and its degradation products (trichloroethanes, dichloroethanes, and vinyl chloride) (Table 2-6). Groundwater is known to contribute to the discharges from SD140, SD170, and SD180 (Table 2-3), but the chemical analyses shown in Table 2-6 indicate that groundwater may be entering SD190 and SD200, as well. Finally, the similarity between the concentrations of these organics in the storm drains and their concentrations in Mitchell Branch several hundred meters downstream is unexpected given their high volatility. This finding suggests that there may be inputs of contaminated groundwater directly to the lower reaches of the stream.

Table 2-6. Mean concentration of tetrachloroethene and its degradation products ($\mu\text{g/L}$) in Mitchell Branch at NPDES monitoring station K-1700 (MIK 0.12) and in storm drains entering Mitchell Branch. Sampling at K-1700 was conducted twice weekly from January through December 1987; storm drains were sampled from March through July 1987 (see Table 2-3). N/A-No data available. Source: NPDES Monthly Reports and Scheib (1987)

	Storm Drain ^a													MIK 0.12
	140	142	144	150	160	162	170	180	190	194	200	210	230	
Tetrachloroethene	<5 (<1-9)	<5	<5	<5	<5	<5	17 (<5-110)	5 (<2-5)	5 (<1-5)	<5	<5	<5	<5 (<5-29)	
1,1-Dichloroethene	<5	<5	<5	<5	<5	<5	6 (<5-18)	5 (<4-6)	<5	<5	<5	<5	N/A	
Trichloroethene	<5	<5	<5	<5	<5	<5	55 (13-330)	50 (9-100)	5 (<2-11)	<5 (<5-21)	13	<5	5 59 (<5-510)	
Trans-1,2-dichloroethene	<5	<5	<5	<5	<5	<5	17 (5-110)	17 (4-34)	17 (<3-39)	<5 (<5-7)	6	<5	N/A	
Vinyl chloride	<10	<10	<10	<10	<10	<10	<10	<10	10 (<5-18)	<10	<10	<10	N/A	

^aStorm drains are numbered sequentially beginning in upper Mitchell Branch and proceeding downstream to MIK 0.12.

Levels of total residual chlorine (TRC) near 1 mg/L were found in SD170 and SD180. These concentrations result from the use of potable water for once-through cooling water, a common industrial practice (Scheib 1987). Although the toxicity of TRC depends upon both the period of exposure and the concentration, the concentrations that were found can be expected to be toxic to many species in Mitchell Branch (Mattice and Zittel 1976). Other contaminants that could be present in storm drains at potentially toxic concentrations include copper (maximum of 77 and 97 $\mu\text{g/L}$ in SD150 and SD230, respectively) and uranium (1,196 $\mu\text{g/L}$ in SD194). Maximum chemical oxygen demand exceeded 30 mg/L in SD140, SD142, SD160, SD162, and SD180. Finally, maximum levels of gross alpha and gross beta each exceeded 200 pCi/L in SD160 and SD194.

2.2.4 Temperature

Temperatures in Mitchell Branch were monitored at MIK 0.50 immediately below the outfall of SD190. Data were collected with a Ryan Tempmentor digital temperature recorder, and values were obtained every 20 min. Monitoring was initiated on April 9, 1987, but the record is incomplete due to equipment problems. Data for the period April-June 1987 are given in Fig. 2-4 and Table 2-7.

Mitchell Branch was characterized by substantially elevated temperatures through the period of record. Mean daily temperatures sometimes exceeded 25°C in Mitchell Branch but were below 20°C in Grassy Creek (Fig. 2-4), a nearby drainage with a similar geology (Sect. 2.1; McMaster 1967, Table 10). Water temperature averaged approximately 4 to 6°C higher in Mitchell Branch than Grassy Creek, although maximum temperatures in the two streams in June differed by as much as 8 to 10°C (Table 2-7). The difference in temperature between the two streams increased from April to June.

The variability in temperature, as indicated by the standard deviation (Table 2-7), also increased over this period but only in Mitchell Branch. In Grassy Creek, water temperatures were less variable in June than in early April, probably due to increasing closure of the riparian canopy. Except for the headwaters, Mitchell Branch above MIK 0.50 has only limited riparian vegetation to moderate water

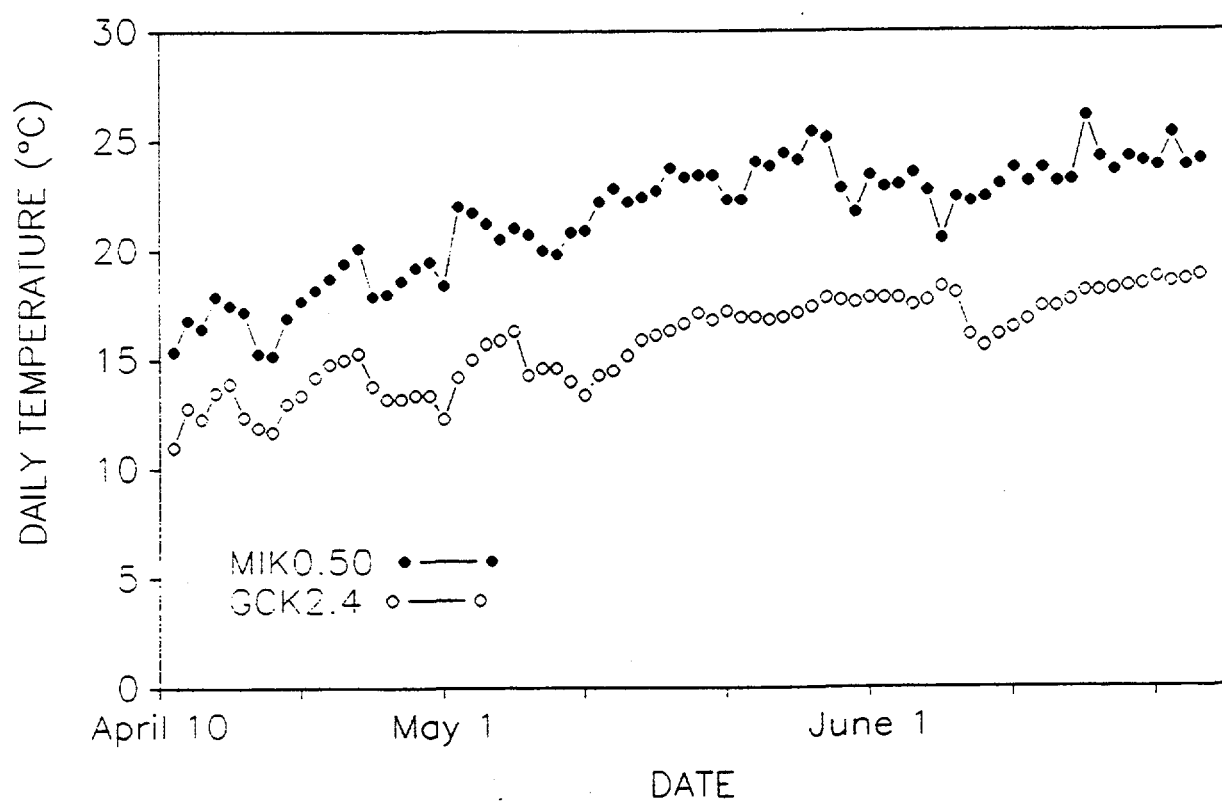


Fig. 2-4. Mean daily water temperature in Mitchell Branch (•) and Grassy Creek (○), a reference stream, April-June 1987.

Table 2-7. Means (standard deviation) and ranges (number of measurements) of water temperature ($^{\circ}\text{C}$) in Mitchell Branch at MIK 0.50 just below the outfall of storm drain 190 and in Grassy Creek at GCK 2.4, a reference site, April-June 1987. Data were obtained with a Ryan Tempmentor digital temperature recorder with values recorded once every 20 minutes

Sampling period	MIK 0.50		GCK 2.4	
	Mean	Range	Mean	Range
April				
Week 2	16.9 (1.9)	12.1-21.4 (432)	12.7 (2.3)	6.8-17.1 (432)
Week 3	17.4 (2.2)	13.1-22.9 (504)	13.4 (2.5)	10.1-19.5 (504)
Week 4	18.9 (1.7)	14.5-23.2 (576)	13.4 (2.2)	8.1-17.9 (576)
May				
Week 1	21.2 (1.7)	17.7-26.0 (504)	15.2 (2.0)	11.4-19.4 (504)
Week 2	21.4 (1.8)	17.4-25.5 (576)	15.0 (2.0)	9.0-18.5 (576)
Week 3	23.0 (1.4)	19.9-26.5 (504)	16.9 (1.0)	14.4-16.9 (504)
Week 4	23.8 (2.4)	18.6-29.6 (648)	17.5 (1.0)	15.8-19.7 (648)
June				
Week 1	22.5 (2.7)	14.7-29.0 (504)	17.1 (1.6)	13.3-19.9 (504)
Week 2	23.5 (3.1)	17.3-29.5 (576)	17.5 (1.2)	14.4-19.4 (576)
Week 3	24.2 (2.2)	20.6-29.2 (504)	18.6 (0.80)	17.4-21.7 (504)

temperatures, which are strongly influenced by both cooling water discharges via SD170, SD180, and SD190 (Table 2-3) and by the retention of effluent in a large open pond (K-1407-B) only 130 m above the monitoring station at MIK 0.50. Warmer air temperatures and the substantially lower stream flow later in the summer (Fig. 2-2) would have resulted in even higher temperatures than the maximum of 29.6°C observed in June.

2.3 BIOLOGICAL MONITORING SITES

Eight sites on Mitchell Branch (Fig. 2-1 and Table 2-8) were routinely sampled to assess the ecological health of the stream. Although the upper site (MIK 1.43) above the ORGDP was the primary reference site, sites on other area streams were also used as secondary reference stations (Fig. 2-2). The lowermost site at MIK 0.12 coincided with the location of the NPDES monitoring station (K-1700). Construction of the weir at MIK 0.12 created a large pool immediately upstream. Mitchell Branch below the weir is also a pool or embayment of Poplar Creek when Watts Bar Reservoir is at full pool (approximately April to October). Water levels in this reach of stream are controlled by operation of Watts Bar Dam on the Tennessee River approximately 61 km below the confluence with the Clinch River. A sampling site for the bioaccumulation task is located at Poplar Creek kilometer (PCK) 6.9 downstream of the mouth of Mitchell Branch (Fig. 2-1).

The remaining five sites were selected based on the location of the three most significant discharges to Mitchell Branch: SD170, SD180 (and effluent from the K-1407-B holding pond) and SD190 (Fig. 2-1). The monitoring sites are located above and below each of these outfalls. Because of the potential for adverse impacts due to construction activities associated with the storage yard immediately northeast of Mitchell Branch, a site (MIK 0.86) was located just downstream of this area and below several storm drains. The sampling sites for the toxicity monitoring and community studies tasks overlap; these tasks/subtasks share four sites, and each includes at least five of the seven primary sites (Table 2-8).

Table 2-8. Location and description of the eight sites on Mitchell Branch that were sampled in various tasks of the ORGDP Biological Monitoring and Abatement Program. Nomenclature of sites (e.g., MIK 0.45) refers to the distance (km) above the confluence of Mitchell Branch with Poplar Creek. NS = Not sampled

Site	Location ^a	Task			
		Toxicity monitoring	Bioaccumulation	Community studies	
				Benthos	Fish
MIK 0.12	NPDES monitoring station K-1700	X	x ^b	NS	NS
MIK 0.45	Below storm drain 190 (45 m)	X	NS	X	X
MIK 0.54	Below K-1407-B effluent and storm drain 180 (90 m) ^c	X	NS	X	X
MIK 0.71	Below storm drain 170 (50 m)	X	NS	X	X
MIK 0.78	Above storm drain 170 (20 m)	NS	NS	X	X
MIK 0.86	Above storm drain 170 (100 m) near storage yard	NS	NS	X	NS
MIK 1.10	Above storm drain 170 (240 m) just downstream of Stoner Road	X	NS	NS	NS
MIK 1.43 ^d	Above storm drain 170 (650 m) and ORGDP	X	e	X	X

^aDistance above/below storm drain is given in parentheses.

^bSampling was also conducted in Poplar Creek at PCK 6.9, approximately 300 m below the mouth of Mitchell Branch.

^cEffluent from the K-1407-B holding pond and storm drain 180 join just below the pond to form a single discharge to Mitchell Branch.

^dReference (control) site.

^eOther reference sites were sampled (see Fig. 2-2).

3. TOXICITY MONITORING

The toxicity monitoring task outlined in the BMAP for Mitchell Branch (Loar et al. 1986b) included three subtasks. The goals of these subtasks were to (1) monitor ambient water toxicity (subtask 1a), (2) measure the toxicity of selected effluents (subtask 1b), and (3) determine point sources of toxicity (subtask 1c). Results of subtasks 1b and 1c are discussed in Sect. 3.1; results of subtask 1a are discussed in Sect. 3.2.

3.1 EFFLUENT TOXICITY

3.1.1 Introduction

The U.S. Environmental Protection Agency (EPA) supports the use of test organisms to estimate the chronic toxicity of effluents and receiving waters (Horning and Weber 1985). As required by the NPDES permit for the ORGDP (EPA 1984), the toxicity of effluents being discharged to Mitchell Branch were evaluated using the 7-day fathead minnow (*Pimephales promelas*) larval survival and growth test, and the 7-day *Ceriodaphnia* survival and reproduction test. These two tests, which are static renewal tests (i.e., the test solutions are replaced daily for each species), are described in detail by Horning and Weber (1985).

The largest discharge to Mitchell Branch is from the K-1407-B holding pond. Because the outfall of this pond is an NPDES monitoring point, this effluent was most extensively tested. Three storm drains (SD170, SD180, and SD190) were selected for toxicity evaluations because (1) they discharged into a reach of Mitchell Branch where biotic communities appeared to be stressed, (2) preliminary toxicity tests indicated that the storm drain effluent markedly reduced the survival of fathead minnows and *Ceriodaphnia*, and (3) they carry the majority of the point and area-source effluents that do not enter the stream via K-1407-B pond [W. J. Scheib, Department of Health, Safety, and Environmental Affairs (HSEA)/ORGDP, personal communication]. The K-1407-B pond discharges at Mitchell Branch kilometer (MIK) 0.63, and

SD170, SD180, and SD190 discharge at MIK 0.76, MIK 0.63, and MIK 0.50, respectively (Fig. 2-1). Sources of water in the storm drains are listed in Table 2-3.

In addition to routine toxicity tests of the effluents, a toxicity reduction evaluation (TRE) was initiated when an effluent was determined to cause a significant reduction in survival or growth of fathead minnows, or survival or reproduction of *Ceriodaphnia*. The TRE was implemented in accordance with Part IV, Sect. 4 of the NPDES permit, which requires a toxicity control plan when the no observed-effect-concentration (NOEC) for the K-1407-B pond was less than 100% (EPA 1984).

3.1.2 Materials and Methods

Chronic toxicity of discharges from the K-1407-B pond, SD170, SD180, and SD190 was estimated with the fathead minnow and *Ceriodaphnia* tests described in Sect. 3.1.1. The K-1407-B pond effluent toxicity was evaluated approximately bimonthly from October 1986 to December 1987. The number of toxicity tests used to evaluate storm drain effluent during the same period was: SD170, 3; SD180, 6; and SD190, 5. SD170 was tested only three times due to construction activities and because high total residual chlorine (TRC) concentrations were sometimes present.

For each toxicity test, daily grab samples were collected at the point of discharge over each 7-day period. A clean 76-L (2 gal.) Nalgene bottle was used to collect and transport the water. Time of collection, water temperature, and arrival time in the lab were recorded. Upon arrival in the laboratory, the water was heated or cooled to 25 °C, and dilutions were made if necessary. Tests with the two species were usually conducted concurrently. On each day of a test, subsamples of each effluent or water sample were analyzed for pH, conductivity, alkalinity, water hardness, TRC, and free chlorine. Other water quality measurements were made by the ORGDP or ORNL analytical laboratories. The water quality parameters listed in this section of the report are those that were consistently above detection limits, had limits included in the NPDES permit (EPA 1984), or which varied considerably from test to test.

Several treatments were used in the effluents TREs. Dechlorination was accomplished by adding 0.1N sodium thiosulfate dropwise into the effluent until the TRC concentration was zero. Filtration through 0.5 μ m pore size glass fiber filters was used to remove particulate matter from the effluent. Aeration for 24 h was used to reduce the concentrations of volatile organics. Evaporation followed by reconstitution in distilled water was used to remove any slightly insoluble materials. This was accomplished by evaporating the effluent to dryness at 90 °C before reconstituting it with distilled deionized water. Peat moss was used as an ion exchanger; this treatment consisted of saturating the peat moss with effluent then filtering the effluent through Nitex netting (80 micron-mesh) to remove coarse particulate matter.

All data analyses were accomplished using the Statistical Analysis System (SAS 1985a, b). Survival percentages for fathead minnow larvae and for *Ceriodaphnia* were transformed (arcsine square-root; Steel and Torrie 1960) before being analyzed statistically. All fecundity values are for females that survived all 7 days of the test. Significant reductions in *Ceriodaphnia* survival and fecundity (compared to a control) were determined using Fisher's Exact Test and Dunnett's Procedure, respectively (Steel and Torrie 1960; Horning and Weber 1985). Significant reductions in fathead minnow survival and growth (compared to a control) were determined using Dunnett's Procedure (Steel and Torrie 1960; Horning and Weber 1985). Dunnett's procedure yields the least significant difference, NOEC and the lowest-observed-effect concentration (LOEC). Unless noted otherwise, statements of significance are based on an $\alpha = 0.05$.

3.1.3 Results

3.1.3.1 K-1407-B Pond

Water chemistry

NPDES data for the K-1407-B pond collected by the HSEA from September 1986 through October 1987 are summarized in Table 3-1. Effluent from the K-1407-B pond exceeded discharge limits during

Table 3-1. Median concentrations and range of selected water quality parameters that were monitored at the K-1407-B pond (NPDES discharge point 003) from September 1986 through October 1987. The median and range were calculated from reported monthly means and ranges. All concentrations are mg/L unless otherwise noted. Source: Oakes et al. (1987) and Rogers et al. (1988)

Parameter	Sample type ^a	Sampling frequency (no./wk)	Median	Range
Aluminum	1	2	0.235	<0.1 - 0.98
Ammonia nitrogen	1	1	<0.2	<0.2 - 0.5
Boron	1	2	0.108	<0.004 - 0.21
Cadmium	1	2	<0.002	<0.002 - 0.05
Chloride	1	1	395.1	105 - 1095
Chloroform ($\mu\text{g/L}$)	2	1	<5	<5 - 51
Chromium	1	2	<0.01	<0.01 - 0.04
COD	1	4	18.24	3 - 91
Copper	1	2	0.007	<0.004 - 0.170
Cyanide	2	1	0.005	<0.002 - 0.016
Dissolved solids	1	4	1490	196 - 3470
Flow (L/min.) ^b	3	7	333	153 - 668
Fluoride	1	4	1.16	<0.1 - 21
Iron	1	2	0.75	<0.05 - 5.02
Lead	1	2	<0.004	<0.004 - 0.052
Magnesium	1	2	21.25	11.0 - 35.0
Manganese	1	2	0.13	0.005 - 1.60
Mercury	1	2	<0.0002	<0.0002 - 0.0003
Nickel	1	2	0.09	0.11 - 1.30
Nitrate-nitrite ^c	1	2	0.8	<0.11 - 22.3
Nitrate nitrogen ^d	1	2	1.92	<0.1 - 26.2
Oil and grease	1	2	2	<2 - 3
Organic carbon (total)	1	2	15.25	2 - 63
pH	3	7	7.75	6.0 - 9.0
PCB (Aroclor-1254) ($\mu\text{g/L}$)	1	1	<1	<1 - 20.4
Phenols (total)	2	1	0.002	<.001 - 0.025
Phosphate	1	1	1.31	0.6 - 431
Silver	1	2	<0.01	<0.01 - 0.014
Sulfate	1	1	564.5	267 - 13402
Surfactants	1	1	<0.20	<0.20 - <0.30
Suspended solids (total)	1	4	12.2	1.0 - 398
Temperature ($^{\circ}\text{C}$)	2	7	19.7	5.5 - 34.4
Total toxic organics	2	1	0.084	0.015 - 0.60
Trichloroethene	2	5	0.006	<0.005 - 0.100
Uranium	1	1	0.10	0.005 - 1.40
Zinc	1	2	0.032	<0.02 - 0.97

^a1 = composite, 2 = grab, 3 = continuous.

^bFlow data were converted from million gallons per day.

^cSeptember 1986 - March 1987 only.

^dApril - October 1987 only.

December 1986 for total suspended solids (TSS) (EPA 1984). The maximum TSS concentration (398 mg/L, December 1986) exceeded the monthly average and daily maximum permit limits of 31 mg/L and 52 mg/L, respectively (EPA 1984). However, the median TSS concentration (12.2 mg/L) was well within permit limits (EPA 1984). Most of the parameters measured did not at any time exceed the EPA water quality criteria for freshwater aquatic life (EPA 1976). However, maximum concentrations of ammonia (0.5 mg/L), cadmium (50 mg/L), cyanide (16 mg/L), and iron (5.0 mg/L) exceeded their respective water quality criteria for freshwater aquatic life of 0.02 mg NH_3 /L, 12 μg Cd/L, 5.0 mg CN/L, and 1.0 mg Fe/L (EPA 1976). The maximum concentration of manganese (1.6 mg/L) exceeded the limit for domestic water supplies (50 μg /L) (EPA 1976).

Water quality measurements of K-1407-B pond effluent obtained during the toxicity tests are summarized in Table 3-2. The effluent was characterized by periods of high conductivity and hardness. From October 1986 to December 1987, mean pH, conductivity, alkalinity, and hardness of full-strength samples ranged from 7.54-8.58, 441-2464 $\mu\text{S}/\text{cm}$, 37.8-142.7 mg/L as CaCO_3 , and 180.0-725.5 mg/L as CaCO_3 , respectively.

Results of some chemical analyses made concurrently with some toxicity tests are summarized in Table 3-3. Concentrations of calcium, chloride, dissolved solids, sodium, and sulfate corresponded with patterns of conductivity and hardness (Table 3-2). The concentrations of most substances did not vary substantially from test to test, with a few notable exceptions: (1) aluminum and iron were elevated in the February test (maximums of 0.75 mg/L and 3.2 mg/L, respectively); (2) magnesium was elevated in the February, March, and April tests (maximums of 26, 26, and 27 mg/L, respectively); (3) nickel was elevated in the March and October tests (maximum values were 1.6 and 1.0 mg/L, respectively); and (4) phosphorus was elevated in the October test (maximum of 4.8 mg/L).

Toxicity tests

From October 1986 to December 1987, effluent from the K-1407-B pond was tested eight times with fathead minnows and ten times with *Ceriodaphnia*. Results of these toxicity tests are summarized in terms

Table 3-2. Water quality measurements of K-1407-B effluent during each toxicity test. Unless otherwise noted, the values for each parameter are the means computed from daily, full-strength samples taken for the indicated 7-day test periods

Test Period	pH	Cond. ^a	Alk. ^b	Hard. ^b	TRC ^c
October 30 - November 6, 1985	8.33	441	142.7	180.0	0.0
December 17-24, 1986	7.82	2260	66.0	725.5	0.0
February 26 - March 5, 1987	7.54	1487	37.8	558.2	0.0
March 26 - April 4, 1987					
untreated	8.33	2464	78.3	598.9	0.0
filtered ^d	8.31	2427	78.7	605.7	0.0
April 23-30, 1987					
untreated	7.95	1968	61.2	701.3	0.0
aerated ^e	8.09	2097	59.2	666.0	0.0
peat moss ^f	5.93	2064	13.3	626.0	0.0
June 18-25, 1987	8.58	1353	55.7	310.0	0.0
July 23-30, 1987	8.53	1919	60.0	320.0	0.0
August 20-27, 1987	7.70	2295	44.1	510.7	0.0
October 22-29, 1987					
untreated	7.90	1639	71.7	435.4	0.0
reconstituted ^g	8.55	1410	17.0	326.0	0.0
December 10-17, 1987					
untreated	8.02	1573	72.1	410.6	0.0
reconstituted ^g	7.71	1386	18.0	272.0	0.0

^aConductivity, $\mu\text{S}/\text{cm}$, corrected to 25 °C.

^bAlkalinity or hardness, mg/L as CaCO_3 .

^cTotal residual chlorine, mg/L.

^dEffluent was filtered through a 0.5 μm pore-size glass-fiber filter.

^eEffluent was aerated for 24 h before being tested.

^fPeat moss was saturated with effluent and then the effluent was filtered before testing.

^gEffluent was evaporated to dryness at 90 °C and reconstituted with deionized distilled water. Values represent measurements on day 1 of the test.

Table 3-3. Selected water quality parameters of effluent from the K-1407-B pond measured during some toxicity tests. Values are the median and range of daily, full-strength samples taken during the indicated 7-day test periods. Concentrations are mg/L. NA = Not analyzed. Source: ORGDP Division of Quality Assurance and Technical Services (unpublished data)

Parameter	Test Period					
	2/26-3/5/87	3/26-4/2/87	4/23-30/87	6/18-25/87	8/20-27/87	10/22-29/87
Aluminum	0.45 (0.28-0.75)	0.235 (0.10-0.27)	0.14 (0.12-0.32)	0.10 (0.064-0.11)	0.19 (0.14-0.29)	0.16 ($<0.1-0.18$)
Boron	0.11 (0.085-0.160)	0.074 (0.049-0.18)	0.13 (0.11-0.15)	0.075 (0.064-0.087)	0.062 (0.024-0.078)	0.014 ($<0.004-0.036$)
Calcium	150 (140-210)	230 (200-290)	260 (210-350)	100 (95-110)	150 (140-300)	150 (140-150)
Chloride	NA	NA	387 (387-387)	194 (176-222)	325 (273-1286)	285 (207-314)
Dissolved solids	NA	NA	1475 (1474-1476)	878 (850-1058)	1200 (1088-3014)	1110 (974-1154)
Iron	2.10 (0.41-3.2)	0.72 (0.42-0.92)	0.38 (0.27-0.52)	0.05 ($<0.004-0.23$)	0.61 (0.21-1.00)	0.36 (0.10-0.51)
Lithium	0.022 (0.014-0.030)	0.0245 (0.021-0.032)	0.032 (0.030-0.041)	0.011 (0.0099-0.014)	0.011 (0.0068-0.014)	0.011 (0.0064-0.014)
Magnesium	20 (17-26)	24.5 (23-26)	25 (25-27)	14 (14-16)	15 (15-18)	13 (13-14)

Table 3-3. (Continued)

Parameter	Test Period					
	2/26-3/5/87	3/26-4/2/87	4/23-30/87	6/18-25/87	8/20-27/87	10/22-29/87
Manganese	0.20 (0.017-0.49)	0.32 (0.15-0.51)	0.10 (0.052-0.18)	0.031 (0.01-0.08)	0.022 (0.0016-0.31)	0.03 (0.01-0.086)
Nickel	0.12 (0.054-0.17)	0.073 (0.051-1.6)	<0.05 (<0.05-<0.05)	<0.05 (<0.05-<0.05)	0.041 (0.022-0.16)	0.35 (0.26-1.0)
Phosphorus	<0.2 (<0.2-<0.2)	<0.2 (<0.2-<0.2)	<0.2 (<0.2-<0.2)	<0.2 (<0.2-<0.2)	<0.2 (<0.2-<0.2)	1.5 (0.28-4.8)
Potassium	18 (13-37)	25 (17-31)	10.0 (8.7-12)	11.0 (9.5-21)	6.3 (5-9.8)	8.1 (7.4-9.0)
Silicon	3.1 (2.1-3.6)	3.2 (2.9-3.6)	3.4 (3.1-3.5)	2.7 (2.6-2.8)	2.8 (2.4-3.5)	2.8 (1.3-3.1)
Sodium	120 (100-170)	345 (330-580)	210 (200-240)	170 (150-220)	160 (150-750)	180 (140-220)
Strontium	0.22 (0.19-0.24)	0.23 (0.21-0.25)	0.23 (0.21-0.27)	0.14 (0.14-0.14)	0.16 (0.14-0.21)	0.15 (0.13-0.15)
Sulfate	NA	NA	NA	NA	NA	330 (283-341)
Suspended solids	9 (8-19)	12 (9-15)	5 (<1-15)	14 (11-20)	13 (8-18)	6 (4-9)

of the effluents' NOEC (Table 3-4). The effluents' NOECs were 100% for fathead minnow survival and growth in six of eight and five of eight tests, respectively; the NOEC was <100% only during the tests conducted in June and August. The effluents' NOEC was 100% for *Ceriodaphnia* survival and reproduction in only two of 10 and three of eight tests, respectively. Mean survival of *Ceriodaphnia* at the end of the 7-day tests was $\leq 40\%$ in seven of 10 tests.

Toxicity reduction evaluations

The water quality measurements and toxicity test results for the TREs conducted on the K-1407-B pond effluent are given in Tables 3-2 and 3-4, respectively. Filtering the effluent through a 0.5 μm pore-size glass-fiber filter or aerating the effluent did not reduce the toxicity of the effluent to *Ceriodaphnia*, but treating the effluent with peat moss (April test) did. The peat moss treatment also markedly reduced the effluent's pH and alkalinity. Evaporating the effluent to dryness and reconstituting it with deionized distilled water reduced the toxicity of the effluent to *Ceriodaphnia* in the October 1987 test, and virtually eliminated the toxicity of the effluent to *Ceriodaphnia* in the December 1987 test. All measured water quality parameters (pH, conductivity, alkalinity, hardness) were lower in the reconstituted effluent than in the untreated effluent. The concentrations of various metals in the untreated and reconstituted K-1407-B pond effluent used in the October test are presented in Table 3-5; these metals were measured by inductively-coupled plasma spectrophotometry. Elements that had notably lower concentrations in the reconstituted effluent (compared to the untreated effluent) included calcium, iron, manganese, nickel, phosphorus, and silicon. More complete chemical analyses of the untreated and reconstituted K-1407-B pond effluent used in the December test are not yet available.

Table 3-4. Summary of toxicity test results with effluent from the K-1407-B pond. Tabular values refer to the effluents' no-observed-effect concentration (NOEC). The NOEC designates the highest tested concentration (%) of the effluent causing no significant ($p > 0.05$) reduction in survival or growth of fathead minnow larvae, or survival or reproduction of *Ceriodaphnia*. NA = no significance test possible due to high mortality

Test Period	FATHEAD MINNOW		CERIODAPHNIA	
	Survival	Growth	Survival	Reproduction
October 30 - November 6, 1986	100%	100%	<100%	<100%
December 17-24, 1986	100%	100%	<20%	<20%
February 2 - March 5, 1987	100%	60%	<20%	<20%
March 26 - April 4, 1987				
untreated	----	----	<50%	<50%
filtered ^a	----	----	<50%	<50%
April 23-30, 1987				
untreated	100%	100%	<100%	NA
aerated ^b	----	----	<50%	NA
peat moss ^c	----	----	50%	50%
June 18-25, 1987	<100%	<100%	100%	100%
July 23-30, 1987	----	----	100%	100%
August 20-27, 1987	50%	<50%	50%	100%
October 22-29, 1987				
untreated	100%	100%	<50%	<50%
reconstituted	----	----	50%	<50%
December 10-17, 1987				
untreated	100%	100%	<100%	NA
reconstituted ^d	----	----	100%	100%

^aEffluent was filtered through a 0.5 μ m pore-size glass-fiber filter before being tested.

^bEffluent was aerated for 24 h before being tested.

^cPeat moss was saturated with effluent and then the effluent was filtered before testing.

^dEffluent was evaporated to dryness at 90 °C and reconstituted with deionized distilled water before being tested.

Table 3-5. Metal analyses (by inductively - coupled plasma spectrophotometry) of untreated and reconstituted effluent from K-1407-B pond tested during October 22-29, 1987. All the values are mg/L. Reconstituted designates full-strength effluent that was evaporated to dryness at 90 °C before being reconstituted with distilled water and tested

Metal	Untreated	Reconstituted
Ag	<0.05	<0.05
Al	<0.2	<0.2
As	<0.1	<0.1
B	<0.0	<0.08
Ba	0.027	0.025
Be	<0.002	0.002
Ca	150	62
Cd	<0.005	<0.005
Co	0.016	<0.01
Cr	<0.04	<0.04
Cu	<0.02	0.027
Fe	0.53	0.081
Ga	<0.3	<0.3
Li	<0.2	<0.2
Mg	16	14
Mn	0.12	0.016
Mo	<0.04	<0.04
Na	170	190
Ni	0.99	0.1
P	6.7	3.9
Pb	<0.2	<0.2
Sb	<0.2	<0.2
Se	<0.2	<0.2
Si	3.5	0.74
Sn	<0.05	<0.05
Sr	0.2	0.1
Ti	<0.02	<0.02
V	<0.02	<0.02
Zn	<0.02	<0.02
Zr	<0.02	<0.02

3.1.3.2 Storm drains

Water chemistry

Water quality measurements of the storm drain effluents taken during the toxicity tests are summarized in Table 3-6. The storm drains were characterized by periods of high concentrations of TRC. The TRC concentrations in SD170, SD180, and SD190 ranged from 0.0-1.70 mg/L, 0.0-0.61 mg/L, and 0.0-1.43 mg/L, respectively.

Results of chemical analyses conducted concurrently with toxicity tests in April and October, 1987 are summarized in Table 3-7. In general, parameter values for SD180 were consistently higher than those for SD190 (i.e. calcium, chloride, dissolved solids, potassium); parameter values for SD170 (analyzed only during the October test) were intermediate to those for SD180 and SD190. During the October 1987 test, SD170 had high concentrations of boron, potassium, and phosphorus relative to SD180 or SD190.

Toxicity tests

Results of the toxicity tests on the storm drain effluents are summarized in Table 3-8. Untreated SD170 effluent had a NOEC of <100% for survival of fathead minnows in two of two tests, and a NOEC of <100% for survival of *Ceriodaphnia* in three of three tests. Untreated SD180 effluent had a NOEC of <100% for survival and growth of fathead minnows in four of five tests and three of four tests, respectively, and a NOEC of <100% for *Ceriodaphnia* survival and reproduction in four of five tests and four of four tests, respectively. Untreated SD190 effluent had a NOEC of <100% for fathead minnow survival and growth in two of four tests and three of four tests, respectively, and a NOEC of <100% for *Ceriodaphnia* survival and reproduction in four of four tests and three of three tests, respectively.

Toxicity reduction evaluations

The water quality measurements and toxicity test results for the TREs on SD170, SD180, and SD190 are summarized in Tables 3-6 and 3-8, respectively. Dechlorinated, full-strength SD170 effluent was not toxic

Table 3-6. Water quality measurements of storm drain effluent taken during the toxicity tests. Unless noted otherwise, the values for each parameter are means computed from daily, full-strength samples taken for the indicated 7-day test periods

Storm Drain	Test Period	pH	Cond. ^a	Alk. ^b	Hard. ^b	TRC ^c
170	10/30-11/6/86	8.15	223	122.3	124.4	0.00
170	08/20-27/87					
	untreated	8.03	304	96.0	134.0	1.70
	dechlorinated ^d	----	---	-----	-----	0.00
170	10/22-29/87					
	untreated	8.07	366	100.4	148.0	1.43
	dechlorinated ^d	----	---	-----	-----	0.00
180	10/30-11/6/86	8.33	441	142.7	145.1	0.10
180	12/11-18/86	8.18	378	133.7	186.0	0.03
180	04/23-30/87	8.23	443	158.0	208.7	0.00
180	08/20-27/87	8.29	344	112.6	163.7	0.61
180	10/22-29/87					
	untreated	8.17	368	122.6	177.4	0.19
	dechlorinated ^d	----	---	-----	-----	0.00
190	10/30-11/6/86	8.22	305	142.7	180.0	0.27
190	04/23-30/87	8.06	512	127.4	178.9	0.00
190	08/20-27/87	8.21	311	106.4	148.3	1.43
190	10/22-29/87					
	untreated	8.05	332	114.3	157.7	0.28
	dechlorinated ^d	----	---	-----	-----	----

^aConductivity, $\mu\text{S}/\text{cm}$, corrected to 25 °C.

^bAlkalinity or hardness, mg/L as CaCO_3 .

^cTotal residual chlorine, mg/L.

^dEffluent was dechlorinated with 0.1N sodium thiosulfate before being tested.

Table 3-7. Selected water quality parameters of storm drain effluent measured during two toxicity tests. Values are medians (and ranges) of daily, full-strength samples taken during the indicated 7-day test period. Concentrations are mg/L. Source: ORGDP Division of Quality Assurance and Technical Services (unpub. data)

Parameter	Test Period				
	4/23-30/87		10/22-29/87		
	SD180	SD190	SD170 ^a	SD180 ^a	SD190
Aluminum	<0.10 (<0.10-0.49)	<0.10 (<0.10-<0.10)	<0.10	<0.10	0.19 (<0.10-0.24)
Boron	0.039 (0.034-0.048)	0.049 (0.041-0.062)	0.32	0.023	0.062 (0.004-0.45)
Calcium	63 (44-69)	46 (37-50)	33	61	42 (31-42)
Chloride	40 ^b (38-42)	17.2 ^b (16.6-17.7)	14	67	15.2 (8.9-15.7)
Diss. Solids	306 (294-318)	230 (228-232)	228	364	206 (166-220)
Iron	0.08 (<0.05-0.60)	0.10 (<0.05-0.47)	0.08	<0.05	0.11 (<0.05-0.20)
Lithium	0.009 (0.005-0.009)	0.010 (0.007-0.012)	0.008	0.007	0.010 (0.007-0.010)
Magnesium	15 (10-15)	13 (9-13)	8.2	11	9.3 (6.9-10.0)
Manganese	2.1 (0.71-2.9)	0.05 (0.04-0.16)	<0.10	<0.10	0.053 (0.043-0.080)
Nickel	<0.05 (<0.05-<0.05)	<0.05 (<0.05-<0.05)	<0.05	0.064	<0.05 (<0.05-<0.05)
Phosphorus	<0.2 (<0.2-<0.2)	<0.2 (<0.2-<0.2)	1.1	0.21	<0.2 (<0.20-0.20)

Table 3-7. (Continued)

Parameter	Test Period				
	4/23-30/87		10/22-29/87		
	SD180	SD190	SD170	SD180	SD190
Potassium	2.8 (2.2-3.0)	3.3 (2.9-4.5)	5.7	2.7	0.96 (<0.60-1.7)
Silicon	3.1 (1.9-3.6)	2.2 (2.0-2.6)	2.2	2.4	1.7 (1.6-3.3)
Sodium	15 (13-17)	12 (8.4-14)	22	37	5.9 (4.5-6.7)
Strontium	0.10 (0.09-0.10)	0.10 (0.08-0.10)	0.076	0.10	0.08 (0.06-0.09)
Susp. Solids	4 (2-34)	9 (<1-30)	<4	4	<4 (<4-9)

^a7-day composite; no range is available.

^bAnalyzed on 4/28 and 4/29 only.

Table 3-8. Summary of toxicity test results with storm drain effluent. Tabular values refer to the no-observed-effect concentration (NOEC) of the effluent. The NOEC designates the highest concentration (%) of the effluent tested causing no significant ($p>0.05$) reduction in survival or growth of fathead minnow larvae, or survival or reproduction of *Ceriodaphnia*. NA = Not applicable due to low survival

Storm Drain	Test Period	Fathead Minnow		Ceriodaphnia	
		Survival	Growth	Survival	Reproduction
170	10/30-11/6/86	<100%	NA	<100%	NA
170	08/20-27/87				
	untreated	-----	-----	<100%	<100%
	dechlorinated ^a	-----	-----	100%	100%
170	10/22-29/87				
	untreated	<100%	NA	<100%	NA
	dechlorinated ^a	100%	100%	100%	<100%
180	10/30-11/6/86	100%	100%	100%	<100%
180	12/11-18/86	<40%	<40%	20%	<20%
180	04/23-30/87	80%	80%	60%	60%
180	08/20-27/87	<40%	<40%	40%	40%
180	10/22-29/87				
	untreated	<100%	NA	<100%	NA
	dechlorinated ^a	100%	<100%	<100%	<100%
190	10/30-11/6/86	<100%	<100%	<100%	NA
190	04/23-30/87	100%	100%	20%	20%
190	08/20-27/87	<20%	<20%	20%	20%
190	10/22-29/87				
	untreated	100%	<100%	<100%	<100%
	dechlorinated ^a	100%	<100%	100%	100%

^aEffluent was dechlorinated with 0.1N sodium thiosulfate before being tested.

to *Ceriodaphnia* in the August test, nor to fathead minnows in the October test. Even after dechlorination, however, SD170 effluent significantly reduced reproduction of *Ceriodaphnia* in the October test. Dechlorinated, full-strength SD180 effluent was not toxic to fathead minnows, but did significantly reduce survival and reproduction of *Ceriodaphnia*. Dechlorinated, full-strength SD190 effluent significantly reduced growth of fathead minnows but was not toxic to *Ceriodaphnia*.

3.1.4 Discussion

3.1.4.1 K-1407-B Pond

Ceriodaphnia were much more sensitive than fathead minnows to K-1407-B pond effluent. A NOEC of <100% for fathead minnow survival occurred in only two of eight tests, while NOEC of <100% for *Ceriodaphnia* survival occurred in eight of ten tests. Similarly, a NOEC of <100% for growth of fathead minnows occurred in three of eight tests, while a NOEC of <100% for *Ceriodaphnia* reproduction occurred in five of eight tests. In addition, the high and sometimes rapid mortality of *Ceriodaphnia* in full-strength K-1407-B pond effluent provides evidence of acute rather than chronic toxicity.

The chemical complexity and daily variation in chemical constituents of the K-1407-B pond effluent makes it difficult to identify the materials contributing to its toxicity. However, by combining good analytical capabilities, a TRE-type approach, and information on toxic concentrations of chemicals to aquatic organisms, some insight was gained into the probable sources of effluent toxicity.

Concentrations of iron and aluminum in the February test were unusually high. The median concentration of iron in the full-strength effluent was 2.1 mg/L and exceeded the concentration of 1 mg/L recommended by the EPA for the protection of freshwater aquatic life (EPA 1976). The reductions in growth of fathead minnows, and survival and reproduction of *Ceriodaphnia* noted during the February test could be attributed, in part, to the high concentrations of these metals.

Fathead minnows exhibited rapid mortality during the June and August tests. The cause of death remains unexplained for the June test;

such rapid mortality (100% mortality within 48 h) was unusual for tests of the pond effluent, and none of the measured parameters were elevated. In the August test, however, the rapid mortality of fathead minnows (100% on the last day of the test) may be partially due to high concentrations of sodium and chloride in the effluent. Results of a fathead minnow toxicity test with pure NaCl in dechlorinated tap water showed that the concentration of NaCl in the pond effluent on the last day of the test was unlikely to entirely account for the observed mortality; the a 48-h LC_{50} for NaCl with fathead minnows is 10.0 g/L (L. A. Kszos and A.J. Stewart, Environmental Sciences Division (ESD)/ORNL, unpublished data).

In general, reductions in survival and reproduction of *Ceriodaphnia* appeared to be linked to high concentrations of hardness in the pond. Excluding the October 1985 test during which the water chemistry was atypical (e.g., lower conductivity and higher alkalinity), significant reductions in *Ceriodaphnia* survival occurred when hardness levels exceeded 320 mg/L. Chemical analyses showed that concentrations of calcium, and magnesium were both elevated during these tests. The TRE completed in December 1987 supports the hypothesis that there is a relationship between mortality of *Ceriodaphnia* and hardness levels in the pond effluent. Untreated effluent had a hardness of 410.5 mg/L and significantly reduced the survival of *Ceriodaphnia*. When reconstituted, the effluent had a hardness of 272 mg/L and did not significantly affect either survival or reproduction of *Ceriodaphnia*.

In the October 1987 test, *Ceriodaphnia* mortality occurred in 50% of full-strength effluent and in the effluent after it was reconstituted. This finding suggests that one or more substances other than calcium contributed to the toxicity. Concentrations of nickel (median = 0.35 mg/L, maximum = 1.0 mg/L) and phosphorus (median = 1.5 mg/L, maximum = 4.8 mg/L) were notably higher during this test than during any other period. Nickel concentrations as low as 95 μ g/L can affect reproduction of *Daphnia magna* (Ewell et al. 1986).

The effect of K-1407-B pond effluent on the biota in Mitchell Branch is certainly dependent on the flow characteristics of the stream

as well as upon its chemical composition. Results of the toxicity tests suggest that the K-1407-B pond effluent may adversely affect the stream biota during periods of low flow in Mitchell Branch. During periods of high flow in Mitchell Branch, greater dilution of the K-1407-B pond effluent would occur, and therefore be less stressful to the biota.

3.1.4.2 Storm drains

High TRC is a major toxicant in the storm drains; in eight of 12 toxicity tests, the TRC concentration was ≥ 0.10 mg/L. Water quality data collected for the storm drains during a 6-wk period in 1987 showed mean TRC concentrations of 0.33 mg/L, 0.175 mg/L, and <0.10 mg/L, in SD170, SD180, and SD190 respectively (Scheib 1987). The EPA water quality criterion for protection of freshwater aquatic life is 0.01 mg/L TRC. Arthur and Eaton (1971) reported a 96-h LC_{50} of 0.22 mg/L TRC for the freshwater crustacean *Gammarus pseudolimnaeus* and found that a 72-h exposure to 0.15 mg/L TRC caused 100% mortality in fathead minnows. The concentrations of TRC measured in the effluents of SD170 and SD190 (maxima of 1.70 mg/L and 1.43 mg/L, respectively) were therefore easily high enough to adversely affect biota in Mitchell Branch.

It is difficult to determine other constituents that contribute to the toxicity of the storm drain effluents when TRC is present in such high concentrations. As indicated by the TREs, removal of TRC by addition of sodium thiosulfate removed toxicity in some, but not all, cases. In addition, there is evidence that sodium thiosulfate is somewhat toxic to *Ceriodaphnia* (A.J. Stewart, ESD/ORNL, unpublished data). Sources of toxicity other than TRC are evident in several tests where the NOEC of the effluent was $<100\%$, even when TRC was not detected (SD170 in October 1986, SD180 and SD190 in April). The results of water quality measurements obtained to date, however, do not point to any specific toxicant(s).

3.2 AMBIENT TOXICITY

3.2.1 Introduction

Ambient toxicity testing was incorporated into the BMAP for Mitchell Branch to (1) evaluate area-source contributions to stream toxicity, (2) characterize patterns of toxicity in Mitchell Branch, (3) document changes in water quality attributable to changes in operations at ORGDP, and (4) provide data to demonstrate that the effluent limitations established for the ORGDP protect and maintain the use of Mitchell Branch for growth and propagation of fish and aquatic life (Loar et al. 1986b). The sites chosen for testing were selected to bracket area- and point-source discharges into the stream, and to correspond closely to those selected as instream monitoring study sites.

3.2.2 Materials and Methods

Ambient toxicity was evaluated using the EPA-approved fathead minnow (*Pimephales promelas*) larval survival and growth test, and the *Ceriodaphnia* survival and reproduction test described by Horning and Weber (1985). These are 7-day static renewal tests that measure the survival and growth of the fathead minnow, and the survival and fecundity of the microcrustacean *Ceriodaphnia*.

The six sites evaluated were located at Mitchell Branch kilometers (MIK) 1.43, 1.0, 0.71, 0.54, 0.45, and 0.12 (Figure 2-1). MIK 1.43 was selected as a reference site because it is located upstream of ORGDP operations and any known source of perturbation. Six tests were conducted on a bimonthly basis from January through November 1987. Water sampling and water chemistry analyses were conducted as described in Sect. 3.1.2. Water from MIK 0.12, however, was collected as a daily 24-h composite for the first four tests and as daily grab samples thereafter. The switch was made to daily grab samples when detectable concentrations of TRC were found in water from MIK 0.12.

All data analyses were accomplished as stated in Sect. 3.1.2 with the following exceptions. Significant reductions in *Ceriodaphnia* survival and reproduction in water from each site were determined in

comparison to survival and reproduction in water from the reference site (MIK 1.43) because survival and fecundity of *Ceriodaphnia* in water from this site were consistently high (mean = 90% and 20.7 offspring/female, respectively). Significant differences in *Ceriodaphnia* survival and reproduction between and within sites were determined using Fisher's Exact Test and the Tukey-Kramer test, respectively (Steel and Torrie 1960; Sokal and Rohlf 1981; Horning and Weber 1985). The Tukey-Kramer test was selected to test for differences in *Ceriodaphnia* reproduction because it accommodates unequal sample sizes whereas Dunnett's test does not (Sokal and Rohlf 1981). Significant differences in fathead minnow survival and growth were determined in comparison to survival and growth in the dechlorinated tap water control that was included with each test. The reference site was not used for such comparisons because fathead minnow survival in water from the reference site was consistently low (mean = 44.4%) and was significantly lower than the control in four of six tests. In comparison, mean survival of fathead minnows in the control water was consistently high ($\geq 87.8\%$). Significant differences in fathead minnow survival and growth were detected using Dunnett's Procedure (Steel and Torrie 1960; Horning and Weber 1985) and the Tukey (for equal sample sizes) or Tukey-Kramer (for unequal sample sizes) tests, respectively (Sokal and Rohlf 1981). Unless otherwise noted, statements of significance are based on an $\alpha = 0.05$.

3.2.3 Results

3.2.3.1 Water Chemistry

Data on water quality parameters measured with each test are summarized in Fig. 3-1. An analysis of variance (ANOVA) indicated that all parameters differed significantly from site to site, and differences between tests were significant for pH, conductivity, and hardness. Several trends were evident: conductivity and hardness increased with distance downstream, and total residual chlorine increased substantially at MIK 0.71, MIK 0.54 and MIK 0.45 before declining to nearly zero by MIK 0.12. The mean pH at MIK 1.0 was significantly higher than at any other site, and the mean pH at MIK 0.12 was significantly lower than at

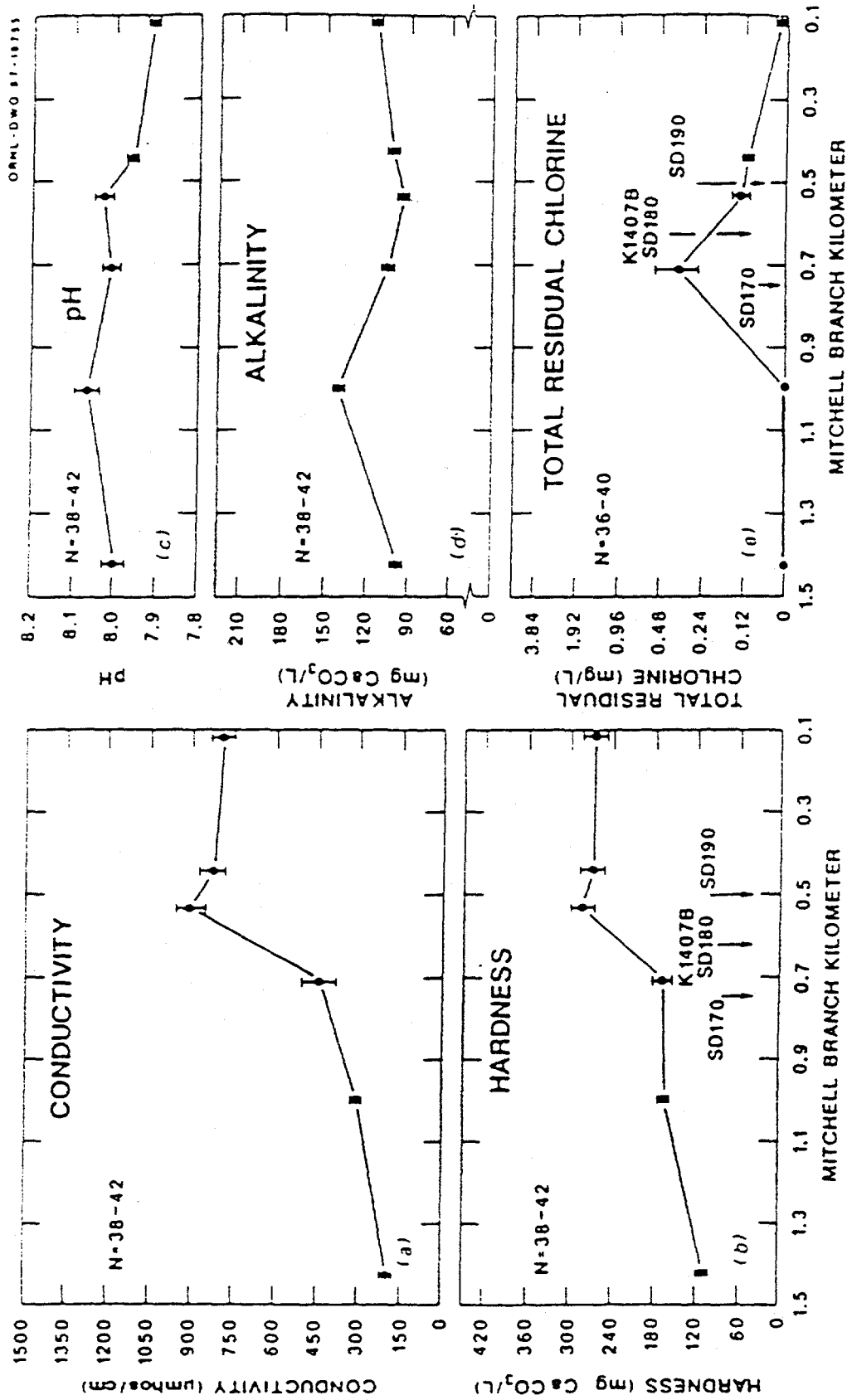


Fig. 3-1. Mean (\pm SE) conductivity, hardness, pH, alkalinity, and total residual chlorine versus distance above the mouth of Mitchell Branch; data from all dates combined. $\mu\text{mhos} = \mu\text{S}$.

all other sites. Mean hardness was not significantly different between the three sites furthest upstream (MIK 1.43, MIK 1.0, and MIK 0.71) or the three sites furthest downstream (MIK 0.54, MIK 0.45, and MIK 0.12). Mean conductivity was not significantly different between the two upstream sites (MIK 1.43 and MIK 1.0), two mid-reach sites (MIK 1.0 and 0.71), or three downstream sites (MIK 0.54 and MIK 0.45 and MIK 0.12). As anticipated, the mean concentration of TRC at MIK 0.71 just below SD170 was significantly higher than that at any other site. No TRC was detected at the two sites above MIK 0.71, but chlorine was usually detected below this site (mean TRC concentrations were not significantly different at the three sites below MIK 0.71).

3.2.3.2 Fathead Minnow Tests

Data on mean survival and growth of fathead minnows at each site are plotted in Fig. 3-2. Differences between sites and tests, and the interaction between site and test were all significant ($P < 0.0001$). Because a sample summary of the entire data set is not possible due to the substantial amount of variation between site and test, the results are discussed in terms of individual tests. Significant reductions in fathead minnow survival and growth for each test are summarized in Table 3-9. Mean survival of fathead minnows in water from MIK 1.43 ranged from 0.0 to 67.5% and was significantly lower than in the control in four of six tests; growth was significantly lower than the control in only one of five tests. Mean survival of the fish in water from MIK 1.0, MIK 0.45, and MIK 0.12 was 75%, 65%, and 85%, respectively, and was significantly lower than the control in one of six tests. Growth of the fish in water from MIK 1.0 and MIK 0.12 was never significantly lower in the test where survival was low. Mean survival of the fish in water from MIK 0.71 ranged from 0.0-2.5% and was significantly lower than the control in four of six tests. Growth of the fish in water from MIK 0.71 could only be evaluated for three tests due to low survival. In these tests, growth of the fish was never significantly lower than the control.

Variation in survival from test to test, expressed as the coefficient of variation (CV), was high for MIK 1.43 and MIK 0.71

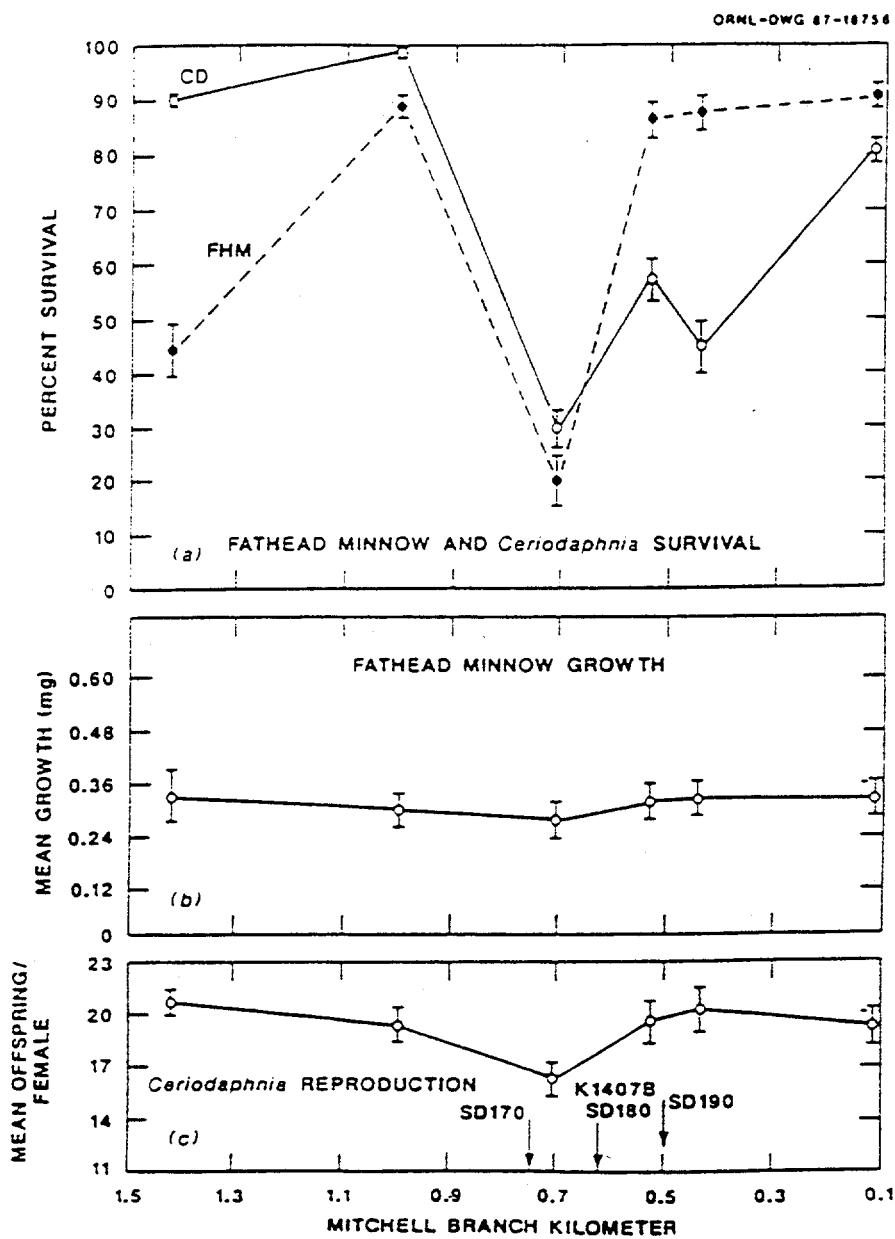


Fig. 3-2. Mean (\pm SE) survival and growth of fathead minnows, and survival and reproduction of *Ceriodaphnia* versus distance above the mouth of Mitchell Branch.

Table 3-9. Comparisons of survival and growth of fathead minnows used in toxicity tests of water from Mitchell Branch. Significant differences from the control (dechlorinated tap water) are shown with an asterisk (*) and no significant difference is shown with a dash (-); NA = no significance test possible due to high mortality

Test Period	Mitchell Branch Kilometer (MIK)					
	1.43	1.0	0.71	0.54	0.45	0.12
1/29-2/5/87						
Survival	*	-	-	-	*	-
Growth	-	-	-	-	*	-
3/26-4/2/87						
Survival	*	*	*	-	-	*
Growth	-	-	-	-	-	-
5/2-9/87						
Survival	*	-	*	-	-	-
Growth	-	-	NA	-	-	-
7/23-30/87						
Survival	-	-	-	-	-	-
Growth	*	-	-	*	-	-
9/17-24/87						
Survival	*	-	*	-	-	-
Growth	NA	-	NA	-	-	-
11/12-19/87						
Survival	-	-	*	-	-	-
Growth	-	-	NA	-	-	-

(74.8% and 141.0%, respectively) compared to the other sites (19.3-29.0%). Based on the six tests, the variation in survival at each site was negatively correlated with mean survival ($r = -0.980$, $N = 6$, $p = 0.0006$). The frequency distribution of mean fathead minnow survival for sites where the CV was low and survival was $>80\%$, and the frequency distribution for sites where the CV was high and survival was $<80\%$, are shown in Fig. 3-3. Although the sample size for the "poor" sites is small ($n = 6$), "good" sites consistently had high survival and "poor" sites such as MIK 0.71 were characterized by both high and low mean survival.

Mean growth of fathead minnows ranged from 0.28 mg/larvae (MIK 0.71) to 0.33 mg/larvae (MIK 1.43, MIK 0.45, and MIK 0.12). Variation (CV) in growth of fish among tests ranged from 42.7% at MIK 0.71 to 70.38% at MIK 1.43. Mean growth and variation in growth of the fish were not correlated ($r = 0.716$, $p = 0.11$).

3.2.3.3 Ceriodaphnia Tests

Mean survival and reproduction of *Ceriodaphnia* at each site are plotted in Fig. 3-2. As for the fish, differences between sites, tests, and the interaction of site and test were all significant ($P < 0.0001$), so results are discussed in terms of individual tests. Results of statistical analyses of survival and reproduction of *Ceriodaphnia* are summarized in Table 3-10. The two sites with the lowest survival were MIK 0.71 and MIK 0.45; survival at these sites was significantly lower than at MIK 1.43 in three of six tests. Reproduction of *Ceriodaphnia* in water from MIK 0.71 and MIK 0.45 was significantly lower than reproduction in water from MIK 1.43 in one of three and one of four tests, respectively. In water from MIK 0.54, survival and reproduction of *Ceriodaphnia* were significantly lower than in water from MIK 1.43 in two of six and one of five tests, respectively. Survival of *Ceriodaphnia* in water from the site furthest downstream (MIK 0.12) was significantly lower than the control in one of six tests, while reproduction was significantly lower than the control in two of six

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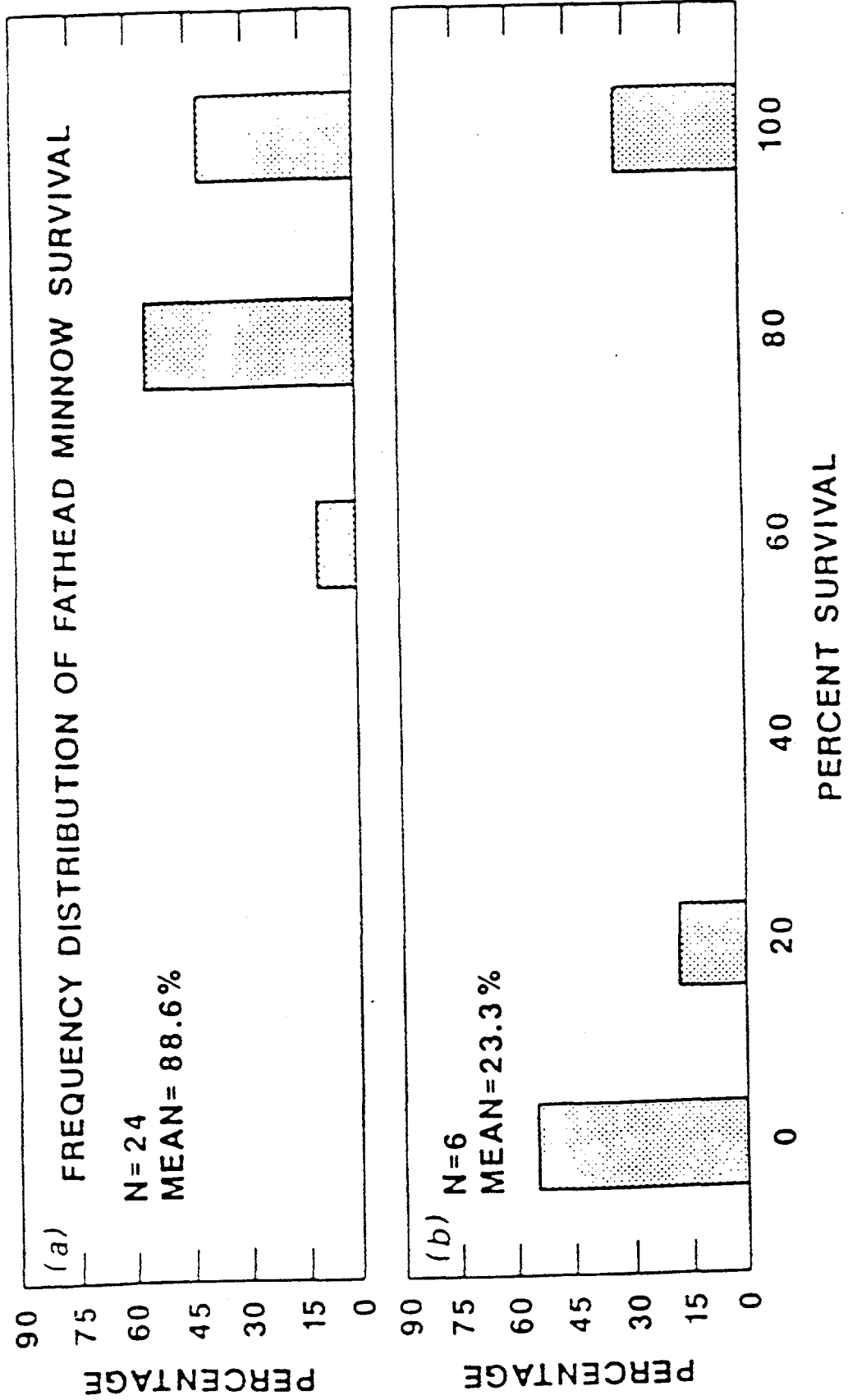


Fig. 3-3. Frequency distribution of arcsin-transformed fathead minnow survival data at (a) MIK 1.0, MIK 0.54, MIK 0.45, and MIK 0.12, and (b) MIK 0.71.

Table 3-10. Comparisons of survival and reproduction of *Ceriodaphnia* in toxicity tests of water from Mitchell Branch. Significant differences from the reference site MIK 1.43, are shown with an asterisk (*) and a lack of significant difference is shown with a dash (-); NA = no significance test possible due to high mortality

Test Period	Mitchell Branch Kilometer (MIK)				
	1.0	0.71	0.54	0.45	0.12
1/29-2/5/87					
Survival	-	-	*	*	-
Reproduction	-	-	-	-	-
3/26-4/2/87					
Survival	-	-	-	*	-
Reproduction	-	-	-	NA	-
5/2-9/87					
Survival	-	*	-	-	-
Reproduction	-	NA	*	*	*
7/23-30/87					
Survival	-	-	-	-	-
Reproduction	*	*	-	-	-
9/17-24/87					
Survival	-	*	-	-	-
Reproduction	-	NA	-	-	-
11/12-19/87					
Survival	-	*	*	*	*
Reproduction	-	NA	NA	NA	*

tests. Water from MIK 1.0 never significantly reduced survival of *Ceriodaphnia* but did significantly affect reproduction in one of six tests.

The CV in *Ceriodaphnia* survival from test to test, ranged from 0.0% (MIK 1.43) to 89.9% (MIK 0.45). Like fathead minnow survival, mean survival of *Ceriodaphnia* was negatively correlated with the variation in survival ($r = 0.967$, $p = 0.0017$, $n = 6$). The frequency distribution of *Ceriodaphnia* survival in sites where the CV was low (and survival was high) versus the frequency distribution of *Ceriodaphnia* survival in sites where CV was high (and survival was low) demonstrated that "good" sites nearly always had high survival and "poor" sites had high survival during some tests but very low survival during others (Fig. 3-4).

Mean reproduction of *Ceriodaphnia* in water from Mitchell Branch ranged from 19.4 (MIK 1.0) to 20.7 (MIK 1.43) offspring/female. The CV in reproduction between sites was low, ranging from 23.7% (MIK 1.43) to 40.7% (MIK 0.54), and was not correlated with mean reproduction ($r = -0.25$, $p = 0.64$).

3.2.4 Discussion

The results of Mitchell Branch ambient toxicity tests showed that the stream is strongly influenced by discharges from the ORGDP. The best example of this influence is the impact of chlorine discharged from the storm drains. Based on a 6-wk survey of the storm drains (Scheib 1987), the mean concentration of TRC in the SD170 discharges was 0.33 mg/L. For the sites on Mitchell Branch, mean survival and reproduction of *Ceriodaphnia* was negatively correlated with TRC ($r = -0.896$, $p = 0.0156$ for survival and $r = -0.828$, $p = 0.0416$ for reproduction). In at least half of the tests with water from MIK 0.71, no fathead minnows or *Ceriodaphnia* survived. Rapid and total mortality is indicative of episodes of acute toxicity, like those due to intermittent releases of chlorine. Patterns of variation in survival support this hypothesis. Sites where mean survival was low did not have consistently poor survival; instead, there were periods of high and periods of very low survival. Sites where mean survival was high did

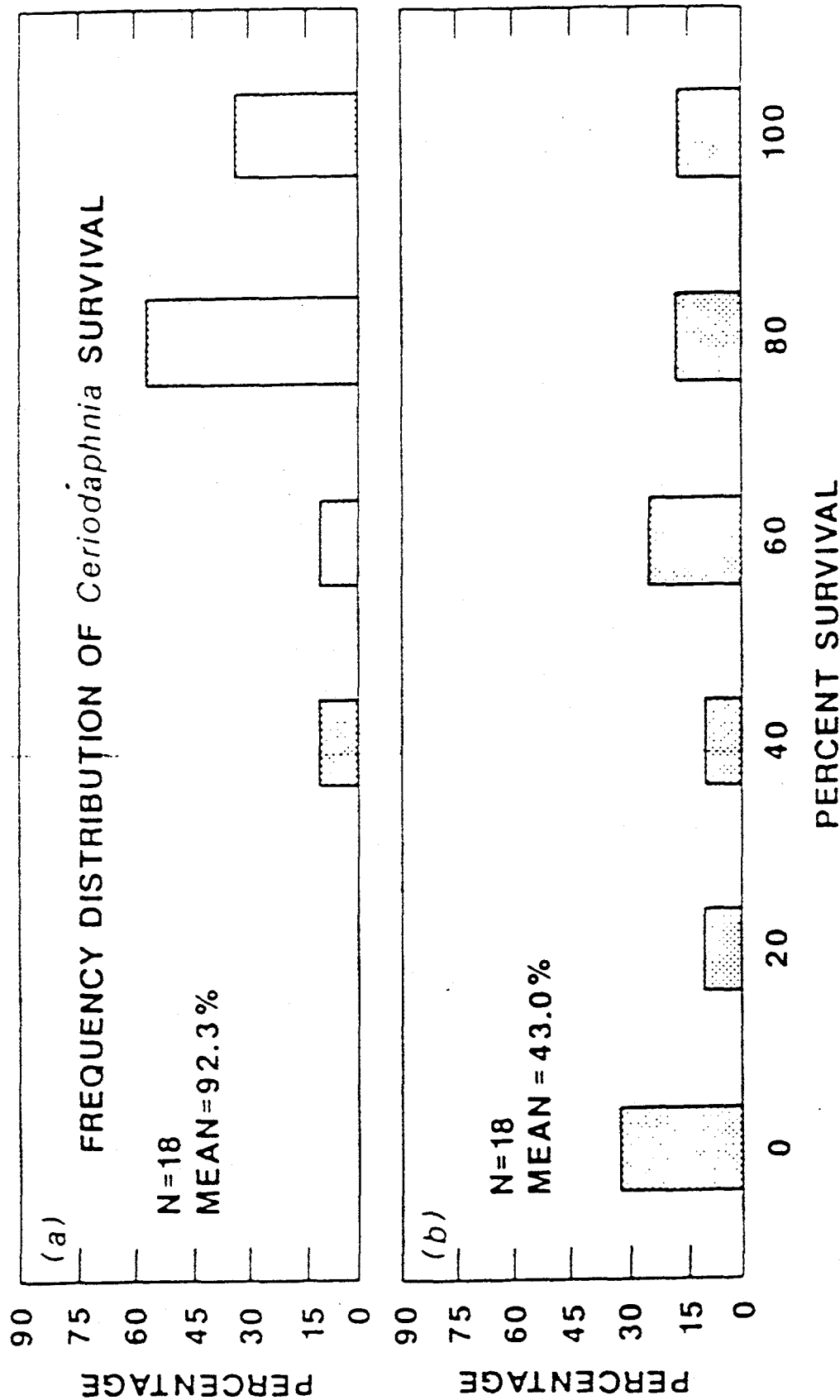


Fig. 3-4. Frequency distribution of arcsin-transformed *Ceriodaphnia* survival data in (a) MIK 1.43, MIK 1.0, and MIK 0.12, and (b) MIK 0.54, MIK 0.45, and MIK 0.71.

not exhibit this variation. Similar patterns have been documented for streams impacted by chlorination at ORNL (Loar et al. 1987).

A second example of the impact of discharges to Mitchell Branch is the marked increase in hardness and conductivity immediately below the discharge of the K-1407-B pond. Comparisons of mean hardness and conductivity in the six sites revealed that the stream could essentially be separated into two reaches. The three sites upstream of the discharge from the K-1407-B pond had low hardness and conductivity and the three sites below the pond were high in hardness and conductivity.

A comparison of toxicity test endpoints showed that only fathead minnow growth and *Ceriodaphnia* reproduction were correlated (Table 3-11). This finding indicates that where acute toxicity is absent, the chronic endpoints for tests with the two species provide similar information. Overall, however, there is little evidence of chronic toxicity in Mitchell Branch; *Ceriodaphnia* reproduction was significantly lower than the control in only five of 24 (21%) test-site combinations, and growth of minnows was significantly lower than the control in only two of 32 (6%) test-site combinations. The lack of correlation between the survival endpoints of the toxicity tests stresses the importance of using more than one species to evaluate ambient toxicity. The sensitivity of the two species to Mitchell Branch water obviously differs, and much more information is gained through the use of both species.

Mean survival of the fathead minnow larvae in water from the uncontaminated reference site (MIK 1.43) was significantly lower than the control in four of six tests. The source of this "toxicity" is currently being investigated. Preliminary data suggest that a bacterial or fungal pathogen may be involved.

3.3 FUTURE STUDIES

The results of effluent and ambient toxicity tests demonstrate the need for continued monitoring of effluents and Mitchell Branch. Monitoring of the storm drains should be continued because they (1) are a major source TRC to the stream, and (2) may contain other toxicants as well (Table 2-5). Efforts will be initiated to document the flow

Table 3-11. Comparison of toxicity test endpoints. FHM = fathead minnow; CD = *Ceriodaphnia*; R = Pearson Correlation coefficient; P is the probability of $|R|$ under the hypothesis $\text{Rho}=0$; an asterisk (*) designates significance at $p<0.05$

Test Endpoint	R	P
CD Reproduction - FHM Growth	0.8658	0.026*
FHM Survival - FHM Growth	0.6671	0.148
CD Survival - CD Reproduction	0.5879	0.220
FHM Survival - CD Reproduction	0.4713	0.345
FHM Survival - CD Survival	0.3486	0.498
CD Survival - FHM Growth	0.3481	0.499

characteristics of each of the storm drains, to evaluate their contribution to in-stream toxicity. Discharge from the K-1407-B pond is expected to be discontinued during 1988. This change will alter the number and sources of discharges monitored, and emphasizes the need for instream monitoring to assess the impacts of such changes on Mitchell Branch. The current schedule of bimonthly testing of ambient toxicity tests and periodic testing of the storm drains will be continued. Toxicity tests will be conducted to evaluate process streams that split from the K-1407-B pond when it is closed.

4. BIOACCUMULATION STUDIES

4.1 INTRODUCTION

Mitchell Branch receives discharges of process water and storm drain effluent as it passes through the ORGDP (Sect. 2.2.1). Metals, hydrophobic organic chemicals, and radionuclides in these discharges (Sect. 2.2) may accumulate in the biota of Mitchell Branch or downstream reaches of Poplar Creek and the Clinch River to levels that may diminish the value of these resources. The primary objectives of contaminant monitoring in Mitchell Branch biota were to (1) identify any substances that accumulate to levels exceeding those observed in biota from nearby, uncontaminated reference streams, and (2) evaluate the extent and significance of contamination by those substances in Mitchell Branch and downstream aquatic systems. Secondary objectives were to assist in locating sources of contaminants that accumulate to unacceptable levels, and to evaluate the relative importance of present vs. past discharges in determining contaminant levels in biota.

4.2 METHODS

In developing the BMAP (Loar et al. 1986b), it was suspected that Mitchell Branch did not support a large enough population of adult game or food fish to meet the sampling requirements of the contaminant monitoring program. This was later confirmed by both quantitative surveys (Sect. 6.2) and separate attempts to collect adult fish. However, the lower reaches of Mitchell Branch (MIK 0.2) did contain adequate numbers of small redbreast sunfish (*Lepomis auritus*) to provide samples for the analysis of metals. Fish were collected in May 1987 for this purpose.

Caged clams (*Corbicula fluminea*) were placed in the stream to monitor for organic contaminants. Clams were obtained in March 1987 from Beaver Creek near Karns in Knox County, Tennessee (Fig. 2-2, lower site), a site previously used to obtain uncontaminated clams for bioaccumulation studies in the Y-12 Plant BMAP (Loar et al. 1988). After holding the clams for 24 h in clean flowing water, they were put

in polypropylene cages and placed in Mitchell Branch in the pool above the NPDES monitoring weir and in the stream/embayment immediately downstream from the weir (Fig. 2-1). Each cage held approximately 50 clams, which contained 0.5-2 g (wet weight) of soft tissue each. The cages remained in the stream for four weeks, after which they were removed and processed prior to delivery to the ORNL Analytical Chemistry Division (ACD) Laboratory for chemical analysis. After the clams were frozen, the shells were removed and the frozen soft tissue placed in a 20-ml glass vial. Composite samples weighing approximately 5 and 10 g each were obtained for polychlorinated biphenyl (PCB) analysis and gas chromatographic/mass spectrometric analysis (GC/MS), respectively. One set of clams from Beaver Creek was frozen immediately for analysis as a control, while another set was held in clean flowing water (dechlorinated tap water) in the laboratory for the duration of the Mitchell Branch exposure and then analyzed as a second control.

When high levels of PCBs were observed in clams placed in Mitchell Branch in March 1987, it was decided to repeat the exposure to evaluate the validity of the results. Clams for this second exposure, which was conducted during July 1987, were obtained from Bull Run in Union County, Tennessee, (Fig. 2-2, upstream site). Results from a comparison of several reference streams conducted in June 1987 as part of the Y-12 Plant BMAP showed that redbreast sunfish and *Corbicula* from this site had the lowest background levels of PCBs of the four streams that were evaluated (G. R. Southworth, ESD/ORNL, unpublished data). In the July exposure clams were held for four weeks in Mitchell Branch in the impoundment above the NPDES weir (the previous exposures showed no difference in PCB accumulation in clams held above vs below the weir). Clams for the controls were frozen at the same time clams were placed in Mitchell Branch and were analyzed for PCBs along with the other samples.

The presence of high levels of PCBs in the Mitchell Branch clam exposures also indicated a need for data to assess possible impacts of this source on PCB levels in fish in adjoining reaches of Poplar Creek. Because East Fork Poplar Creek (EFPC) enters Poplar Creek a short distance upstream and has been shown to be a source of PCBs to Poplar Creek, fish were collected from Poplar Creek immediately downstream of

Mitchell Branch. This collection was coordinated with the routine monitoring of PCBs in fish from lower EFPC, as stipulated in the Y-12 Plant BMAP (Loar et al. 1985a), in order to facilitate comparisons of PCB levels in fish above and below the mouth of Mitchell Branch. Bluegill (*Lepomis macrochirus*) and redbreast sunfish (*L. auritus*) were collected by electrofishing at Poplar Creek kilometer (PCK) 6.9, a reach approximately 400 m long starting 200 m downstream from the mouth of Mitchell Branch (Fig. 2-1). Reference fish were obtained from Hinds Creek in Anderson County, Tennessee (Fig. 2-1).

Fish collected at each site were placed on ice in a labelled ice chest and returned to the laboratory for processing. Upon return to the laboratory, fish were tagged with a unique four-digit tag wired to the lower jaw. Each fish was then weighed and measured, and scale samples were taken for age determination. The fish were fileted and skinned. On small fish collected in Mitchell Branch, a 1-2 g portion of the anterior dorsal axial muscle filet was excised for the determination of mercury, and the remainder of the filets was used for analysis of other metals. Samples were wrapped in heavy-duty aluminum foil, labelled, and stored at -20°C in a locked freezer in Building 1504 at ORNL until delivered to the ACD/ORNL Laboratory for analysis.

Mercury determinations were carried out by ACD using procedure EC 420 (Martin Marietta Energy Systems 1983). Samples were digested in a mixture of nitric acid, perchloric acid, and potassium dichromate, after which the mercury was reduced with stannous chloride and determined by cold vapor atomic adsorption spectrophotometry. Other metals were determined using graphite furnace atomic adsorption spectrophotometry following digestion with concentrated nitric acid (EPA 1980). PCBs were determined by packed column gas chromatography following methylene chloride extraction and adsorption column cleanup (EPA 1980). Organic priority pollutants were analyzed by procedure PPB 12/83 (EPA 1983), in which the homogenized sample is extracted in methylene chloride, cleaned up using column chromatography, and analyzed using capillary column gas chromatography with mass spectrometric or electron capture detectors.

Statistical evaluations of data were made using SAS procedures and software (SAS 1985a,b) for analysis of variance, Duncan's multiple range test, linear regression analysis, and the calculation of means, standard deviations, standard errors, and coefficients of variation.

Quality assurance was maintained using a combination of (1) blind duplicate analyses; (2) split sample analyses between the EPA Environmental Services Laboratory, Athens, Georgia, and the ACD Laboratory at ORNL; (3) and the analyses of biological reference standards and uncontaminated fish. Recoveries of PCBs were verified by spiking uncontaminated fish or clam samples with known amounts of PCBs and analyzing them. Details and results of QA analyses are summarized in Appendix A and results of tissue analyses are tabulated in Appendix B.

4.3 RESULTS

4.3.1 Metals in Mitchell Branch Fish

Analyses of sediment (samples taken at various points in Mitchell Branch) in 1986 by Ashwood et al. (1986), indicated that the stream contained elevated levels of a number of metals. Based on a comparison with other streams sampled within the ORGDP area, Ashwood et al. concluded that arsenic, cadmium, chromium, copper, mercury, nickel, lead, selenium, silver, and zinc were abnormally high. Comparison of those data with similar analyses of sediments from several offsite reference streams that were sampled in 1987 as part of the Y-12 Plant BMAP confirmed their conclusion (Table 4-1). Copper, mercury and nickel are present in Mitchell Branch sediments at concentrations more than 10 times higher than the levels in these reference streams; zinc is also strikingly elevated.

Despite the presence of abnormal levels of metals in sediments, concentrations of metals in fish collected from Mitchell Branch were similar to those in fish from Hinds Creek, a reference stream (Table 4-2, Appendix B, Table B-1). Although cadmium, copper, and mercury were all slightly higher in Mitchell Branch fish, only the

Table 4-1. Comparison of metals from sediments in Mitchell Branch and reference streams. Tabular values are mean \pm SE in ppm dry wt

Metal	Mitchell Branch ^a (n=13)	Reference streams ^b (n=4)
Arsenic	52 \pm 16	<19
Cadmium	1.3 \pm 0.3	1.0 ^c
Chromium	77 \pm 7	42 \pm 4
Copper	268 \pm 25	17 \pm 0.5
Lead	77 \pm 9	58 \pm 6
Mercury	40 \pm 0.7	0.04 \pm 0.01
Nickel	745 \pm 96	19 \pm 0.6
Selenium	53 \pm 23	<37
Silver	7.8 \pm 6.8	<9.5
Zinc	300 \pm 31	98 \pm 13

^aSamples were collected at various points along Mitchell Branch.
Source: Ashwood et al. 1986.

^bDepositional surface sediments were collected in July 1987 from the following reference streams (See Fig. 2-2): Beaver Creek (upper site), Brushy Fork, Bull Run (lower site) and Hinds Creek (G. R. Southworth, ESD/ORNL, unpublished data).

^cDetection limit was <0.9 ppm.

Table 4-2. Metal concentrations in fish (mean \pm SE in ppm wet wt) from Mitchell Branch and Hinds Creek (reference stream). Mitchell Branch fish were immature redbreast sunfish (*Lepomis auritus*, n = 6) and adult creek chub (*Semotilus atromaculatus*, n = 1); Hinds Creek fish were adult redbreast sunfish (n = 1) and bluegill (*Lepomis macrochirus*, n = 3)

Metal	Site		PGV ^a
	Mitchell Branch	Hinds Creek	
Antimony	< 0.2	< 0.3	5.2
Arsenic	< 0.10	< 0.10	0.0007
Beryllium	< 0.05	< 0.05	0.004
Cadmium	0.05 \pm 0.01	0.015 \pm 0.007	1.0
Chromium	< 0.1	< 0.1	1.8
Copper	0.64 \pm 0.24	0.45 \pm 0.28	36
Lithium	0.5	< 0.5	--
Mercury	0.19 \pm 0.02	0.09 \pm 0.03 ^b	0.42 ^c
Nickel	< 1.0	< 1.0	5.2
Lead	0.03 \pm 0.003	< 0.02	1.8
Selenium	0.55 \pm 0.10	0.55 \pm 0.06	12
Silver	< 0.05	< 0.10	0.29
Thallium	< 0.2	< 0.2	0.66
Zinc	6.4 \pm 1.0	6.2 \pm 0.5	180

^aPreliminary Guidance Values (from Travis et al. 1986, Hoffman et al. 1984).

^bAdult redbreast sunfish, Hinds Creek, n = 20.

^cFDA action level is 1.0 ppm.

difference in mercury concentration in fish from the two streams was statistically significant ($\alpha = 0.05$).

The levels of metals in Mitchell Branch fish were also quite similar to those observed by the Tennessee Valley Authority (TVA) at one of their reference sites (Melton Hill Reservoir) (TVA 1985, 1986). Moreover, the levels of metals observed in fish in the National Contaminant Biomonitoring Program (geometric mean of 112 sites sampled for arsenic, cadmium, copper, lead, mercury, selenium, and zinc; Lowe et al. 1985) were also generally similar to levels observed in Mitchell Branch fish, except mercury. A comparison of the concentrations of metals in Mitchell Branch fish with Preliminary Guidance Values (PGVs), derived to screen for levels of contamination that may potentially threaten human health (Travis et al. 1986; Hoffman et al. 1984),

indicates that only arsenic, beryllium, and mercury approach this threshold (Table 4-2). Neither arsenic nor beryllium is shown to be elevated by these data; however, the PGV is set at a level below background due to the carcinogenicity of these two metals. The PGV screening approach is very conservative and is designed to eliminate from concern any substances not exceeding the PGV (Hoffman et al. 1984).

Although mercury was clearly elevated in fish from Mitchell Branch relative to the reference stream fish, the difference was not large and levels were well below the FDA action level of 1.0 ppm (FDA 1984a). However, mercury exhibits a tendency to accumulate to higher levels as a function of the duration of exposure (age) of fish; therefore, immature fish such as those collected in Mitchell Branch would be expected to contain lower levels of mercury than adult fish from the same site (Elwood 1984). In order to adjust for this, the relationship between fish weight and mercury concentration observed in fish from lower Poplar Creek in 1976 by Elwood (1984) was used to calculate the level of mercury that would be expected in adult sunfish at this site. In that study, small bluegill (~5 g) contained approximately 0.15 ppm mercury at a site where adult fish contained approximately 0.4 ppm. The mean concentration of mercury in immature redbreast sunfish from Mitchell Branch (0.17 ppm) can be used to infer a level of 0.3-0.5 ppm in adult fish (if they were present). This estimate is quite close to the level

of mercury (0.3-0.4 ppm) observed in adult redbreast sunfish and bluegill from Poplar Creek in the vicinity of Mitchell Branch (Sect. 4.3.2). This result suggests that either immature redbreast sunfish collected in Mitchell Branch were migrants from nearby reaches of Poplar Creek, where they had accumulated elevated levels of mercury, or that Mitchell Branch is the source of abnormal mercury levels in these fish. The presence of 0.29 ppm mercury in the single creek chub (*Semotilus atromaculatus*), a species found primarily in small streams and unlikely to have come from Poplar Creek, suggests the latter; however, there are no measurements of mercury in creek chubs from local reference streams with which to compare this value. Results of routine NPDES water chemistry monitoring and the 1987 storm drain survey (Table 2-5) also indicate that Mitchell Branch is a source of mercury contamination to fish.

4.3.2 Mercury in Poplar Creek Fish

Mercury was measured in adult sunfish from reaches of Poplar Creek downstream of Mitchell Branch to evaluate a possible contribution of mercury from Mitchell Branch. Results of these analyses, and comparison with analyses of fish from lower EFPC, which enters Poplar Creek approximately 1.5 km upstream of Mitchell Branch are presented in Table 4-3. Because no relationship was observed between mercury concentration in fish tissue and fish weight ($\alpha = 0.05$), statistical comparisons of mean mercury levels could be made without adjusting for differences in fish size among the two sites. By collecting only adult sunfish, the variation in mercury levels with weight was minimized, a protocol that has been utilized previously in the Y-12 Plant BMAP for EFPC. The mean levels of mercury did not differ significantly ($\alpha = 0.05$) between lower EFPC and Poplar Creek among either bluegill or redbreast sunfish. Levels of mercury in sunfish in EFPC decreased nearly in proportion to dilution of the New Hope Pond discharge at sites along the length of the stream (G. R. Southworth, ESD/ORNL, unpublished data). After entering Poplar Creek, water from EFPC is diluted nearly four-fold (Loar et al. 1981b). However, mercury concentrations in fish at the Poplar Creek site are not lower than those in fish from lower

Table 4-3. Concentrations of Hg and PCBs (mean \pm SE in ppm wet wt) in bluegill (*Lepomis macrochirus*) and redbreast sunfish (*L. auritus*) upstream and downstream of the mouth of Mitchell Branch, June 1987

Site	Species	Hg	PCB	N
EFK 2.1 ^a	bluegill	0.38 \pm 0.08	0.22 \pm 0.06	9
PCK 6.9 ^b	bluegill	0.42 \pm 0.05	0.17 \pm 0.06	8
EFK 2.1	redbreast	0.47 \pm 0.09	0.35 \pm 0.06	7
PCK 6.9	redbreast	0.28 \pm 0.08 ^c	0.22 \pm 0.06	4
Hinds Cr. (Reference)	bluegill	0.06 \pm 0.01 ^d	0.03 \pm 0.01	6
Hinds Cr. (Reference)	redbreast	0.09 \pm 0.03 ^d	0.06 \pm 0.02	6

^aEast Fork Poplar Creek kilometer (EFK) 2.1.

^bPoplar Creek kilometer (PCK) 6.9.

^cN = 3.

^dN = 20, May 1985 - June 1987.

EFPC. It appears that either changes occur in the environmental chemistry of mercury in lower Poplar Creek, enhancing the biological availability of water-borne or sediment-associated mercury, or additional sources of mercury occur in the reach of Poplar Creek near the ORGDP.

The levels of mercury observed in bluegill from Poplar Creek near the mouth of Mitchell Branch in 1987 were virtually the same as levels observed in 1976-1977 (Loar et al. 1981b; Elwood 1984) and 1984 (TVA 1985). Apparently little or no change in the degree of mercury contamination at this site has occurred in the past decade.

4.3.3 Organic Contaminants

Results of analyses of caged clams for organic priority contaminants by GC/MS did not reveal the presence of significant contamination with a large number of organic compounds. Only two compounds other than PCBs were detected, di-n-butylphthalate and bis (2-ethylhexyl)phthalate (Table 4-4, Appendix B, Table B-4). Names and detection limits of other organic compounds that were included in the screening analyses but were not detected are listed in Appendix B, Table B-5. The presence of detectable levels of phthalates in clam samples is consistent with the presence of these two compounds in storm drains discharging to Mitchell Branch (Scheib 1987) and their moderate bioaccumulation potential (Callahan et al. 1979). Unlike PCBs and some other chlorinated hydrophobic chemicals, phthalates do not accumulate in aquatic biota (especially fish) to levels that are several orders of magnitude higher than levels found in the water to which the biota are exposed (Callahan et al. 1979). Consequently the presence of these compounds in clams would not lead to the inference that fish living in Mitchell Branch would accumulate similar levels.

The detection of PCB-1254 and PCB-1260 in clam samples in the organics screening analysis corroborates the identification of these two PCB mixtures in clam samples that were analyzed by packed column gas chromatography, a technique used to quantify low levels of PCBs in aquatic organisms (Section 4.3.4). Concentrations reported in the screening analysis were somewhat lower than those reported in the packed

Table 4-4. Concentrations of organic compounds in caged Asiatic clams (*Corbicula fluminea*) maintained for four weeks in Mitchell Branch, March 18 - April 15, 1987

Compound ^a	Concentration (ppm wet wt)		Occurrence ^b	Control
	Mean \pm SD	Range		
PCB-1254	1.9 \pm 1.3	0.3 - 3.5	5/5	<0.1
PCB-1260	0.14 \pm 0.05	<0.1 - 0.22	3/5	<0.1
Total PCB	2.0 \pm 1.3	0.57 - 3.7	5/5	<0.2
Di-n-butylphthalate	6.1, 9.0 ^c	<2.0 - 9.0	2/5	<2.0
bis (2-ethylhexyl)-phthalate	4.2, 6.4 ^c	<2.0 - 6.4	2/5	<2.0

^aNames and detection limits of other compounds that were screened for but not detected are listed in Appendix B, Table B-5.

^bNumber of samples exceeding detection limit/ total number of samples.

^cActual values for the two samples that contained detectable levels.

column gas chromatographic analysis (Table 4-5), but are nevertheless indicative of substantial PCB contamination.

4.3.4 Polychlorinated Biphenyls (PCBs)

Detectable levels of PCBs were measured in the effluent of the K-1407-B pond in January through April 1987 (Rogers et al. 1988). The source of PCBs to the pond was located and eliminated, and PCB levels in the pond outfall have remained below the detection limit of 1 $\mu\text{g/L}$ since April 1987 (Bill Scheib, HSEA/ORGDP, personal communication).

4.3.4.1 PCBs in Caged Clams

Asiatic clams (*Corbicula fluminea*) held for four-week exposures in cages in Mitchell Branch accumulated substantial concentrations of PCBs (Tables 4-5 and B-6). Clams exposed to Mitchell Branch in March-April 1987 accumulated 3.9 ppm PCBs; those exposed in July-August 1987 accumulated 2.5 ppm. In both cases, virtually all the PCB was reported as Arochlor 1254. In order to place these results in perspective, they were compared with the analyses of clams maintained in a similar fashion in EFPC immediately below the outfall of New Hope Pond (EFK 23.7) (G. R. Southworth, ESD/ORNL unpublished data). As shown in Table 4-5, the EFPC clams accumulated substantially lower concentrations of PCBs, averaging about 0.5 ppm. Redbreast sunfish collected at EFK 23.7 contained approximately 1.1 ppm PCBs. A PCB exposure that can produce 1 ppm levels in low-lipid fish, such as redbreast sunfish, would undoubtedly result in levels in excess of the 2 ppm FDA action level (FDA 1984b) in older, fattier, fish such as carp (*Cyprinus carpio*), channel catfish (*Ictalurus punctatus*), and old largemouth bass (*Micropterus salmoides*). The clam data suggest that PCB exposure in Mitchell Branch is probably high enough to produce levels of contamination in adult sunfish in excess of the FDA limit.

While PCB contamination of fish within Mitchell Branch causes no concern for human exposure due to the absence of fish of edible size in that system, the high levels of PCBs observed in clams indicate that the stream may be a significant source of contamination to Poplar Creek. If the concentration of PCBs found in clams after a four-week exposure is

Table 4-5. Mean (\pm SE) concentrations of PCBs in caged Asiatic clams (*Corbicula fluminea*) maintained for four weeks in Mitchell Branch and East Fork Poplar Creek

Site	Exposure Period	Concentration (ppm, wet wt)	N	Control
MIK 0.14 ^a	3/18 - 4/15/87	3.9 \pm 0.3	5	0.28 ^b
MIK 0.14	7/15 - 8/17/87	2.5 \pm 0.1	3	0.05 ^c
EFK 23.4 ^d	7/7 - 8/6/86	0.45 \pm 0.04	3	0.07
EFK 23.4	4/28 - 5/26/87	0.57 \pm 0.11	3	0.08
EFK 23.4	9/1 - 9/29/87	0.40 \pm 0.06	3	0.03

^aMitchell Branch, kilometer MIK 0.14.

^bBeaver Creek (lower site, Fig. 2-2).

^cBull Run (upper site, Fig. 2-2).

^dEast Fork Poplar Creek kilometer (EFK) 23.4.

assumed to be proportional to the concentration of PCBs in the water column, then the ratio of PCB levels in Mitchell Branch clams to EFPC clams (6.8) is equal to the ratio of PCB concentrations in the two systems. By multiplying the mean annual flow in Mitchell Branch for 1985-1987 (12.9 L/S, Table 2-1) by the numerator of this ratio (6.8), and comparing it with the product of the mean annual flow for 1985-1987 in EFPC at New Hope Pond (394 L/S, Table 2-1) and the denominator of the ratio (1), it is possible to estimate the relative contributions of Mitchell Branch and EFPC to PCB contamination in lower Poplar Creek. This analysis yields values of 88 and 394 for Mitchell Branch and EFPC respectively, indicating that while Mitchell Branch may be (or may have been) a significant contributor to PCB contamination in biota of lower Poplar Creek, the input of PCBs via EFPC is substantially more important.

4.3.4.2 PCBs in Poplar Creek fish

The results of PCB analyses of sunfish collected upstream and downstream of the mouth of Mitchell Branch (Table 4-3) provide another opportunity to discern the impact of PCB inputs from that source. As was noted for mercury, there was no statistically significant relationship between PCB concentrations in fish and fish weight ($\alpha = 0.05$). Redbreast sunfish collected downstream of the mouth of Mitchell Branch contained significantly higher levels of PCBs ($\alpha = 0.05$) than redbreast sunfish collected from lower EFPC. Although bluegill contained a slightly higher mean PCB concentration at the downstream site, the means of the two sites were not significantly different ($\alpha = 0.05$). As noted previously, the flow of EFPC is diluted about four-fold by Poplar Creek before reaching the mouth of Mitchell Branch; therefore, a decrease in PCB levels in fish would be expected. While not conclusive, these data support the hypothesis that Mitchell Branch was a significant source of PCB contamination to lower Poplar Creek, in 1987.

Although, the levels of PCBs observed in sunfish from lower Poplar Creek (~0.2 ppm) exceeded those found in sunfish from a noncontaminated reference stream (~0.04 ppm), the degree of contamination would not be

considered severe. Carp, catfish and other long-lived, oily fish residing year-round in this reach of Poplar Creek would likely attain PCB concentrations of 1-2 ppm, and a significant number of fish would exceed the FDA limit. Bluegill collected in 1977 from lower Poplar Creek contained approximately 0.4 ppm PCBs (Loar et al. 1981b), indicating that fish in this reach have contained PCB residues for at least the past 10 years.

4.4 CONCLUSIONS

The data collected in 1987 indicate that at least two contaminants (mercury and PCBs) appear to accumulate to above background levels in Mitchell Branch biota. The levels of mercury are not excessive relative to the FDA limit, and are not atypical of the degree of contamination found in fish in nearby Poplar Creek. However, the significance of Mitchell Branch as a source of the elevated mercury levels observed in fish from lower Poplar Creek cannot be ascertained from these data. Levels of other metals in fish from Mitchell Branch were not of concern from the standpoint of bioaccumulation.

Unlike mercury, the PCB monitoring data are more conclusive, clearly documenting that Mitchell Branch is a source of PCB contamination to its biota, and suggesting that the stream may also be a significant source of PCB contamination to biota in lower Poplar Creek. On two separate occasions, the levels of PCBs in caged clams held in Mitchell Branch were well above the 2 ppm FDA limit after only a one-month exposure; most fish inhabiting this system would likely also exceed the FDA limit. Even though levels of PCB contamination in Poplar Creek sunfish, were well below the FDA limit, they were high enough to suggest that other species, such as channel catfish, that have a higher propensity to accumulate PCBs, could contain in excess of 2 ppm PCBs if they were permanent residents of lower Poplar Creek.

The identification and removal of a recently active source of PCBs to Mitchell Branch may have eliminated or greatly reduced its significance as a continuing source of PCBs to Poplar Creek. However, residual contamination in surface sediments may continue to release PCBs

to the water column and maintain Mitchell Branch as an ongoing source of contamination to biota for some period of time.

Phthalates were also detected in some clam samples from Mitchell Branch. It is likely that there are sources of phthalate contamination to Mitchell Branch, including SD180 and SD190 (Scheib 1987) and the K-1407-B pond (Oakes et al. 1987, Table 5.3.51). These data must be regarded with some caution, however, since phthalates are a common laboratory and sampling contaminant and only a portion of the samples contained detectable levels.

4.5 FUTURE STUDIES

The most significant contaminant bioaccumulation problem associated with Mitchell Branch appears to be PCBs. A key objective of future studies will be to develop a better understanding of the role of Mitchell Branch as a source of PCBs to lower Poplar Creek, and to identify and characterize the PCB source(s) in Mitchell Branch (e.g. specific ongoing discharges, sediment contamination, episodic releases, etc).

Assessment of the accumulation of radionuclides (uranium-235,238, technetium-99) in Mitchell Branch biota requires larger samples than were obtained in the 1987 collection, which was limited by the depauperate fish fauna of Mitchell Branch. If adult fish cannot be collected in early 1988 from this system, caged fish will be introduced and monitored in conjunction with the biological indicator studies (Sect. 5.0). The use of small resident fish and caged clams for monitoring the accumulation of metals and organics will be continued, unless larger fish can be obtained. Continued monitoring should also provide more insight into the significance of phthalate contamination in Mitchell Branch.

5. BIOLOGICAL INDICATOR STUDIES

This component of the ORGDP BMAP involves the application of biological indicators for evaluating the effects of water quality on the physiological health of fish populations in Mitchell Branch. A suite of biochemical, physiological, and organism-level indicators have been selected to determine the level of both short and long-term stresses on these populations.

The first year of study was designed to (1) measure a wide variety of bioindicators representing several levels of biological organization, and (2) select a subset of those indicators that provided the most reliable information on the response of fish to water quality in Mitchell Branch. Both quantitative studies (Sect. 6.2) and other efforts to collect fish (Sect. 4.2) indicated that there were insufficient numbers or sizes of fish in the stream to accomplish this task. During the past two years however, similar types of bioindicator studies have been conducted in streams on the DOE Oak Ridge Reservation including White Oak Creek (WOC) and East Fork Poplar Creek (EFPC) (Loar et al. 1987; Loar et al. 1988). In both of these systems, a wide variety of biochemical, physiological, and organism-level indicators were measured and a subset of these indicators was identified for seasonal biomonitoring studies (Table 5-1). It is reasonable to assume that these same indicators could also be measured on fish that could be stocked in Mitchell Branch because (1) the same set of indicators was found to represent the response of bluegill (*Lipomis macrochirus*) and redbreast sunfish (*L. auritus*) to chronic contaminant stress in both WOC and EFPC, and (2) hybrid sunfish [bluegill x green sunfish (*L. cyanellus*)] whose physiology is probably very similar to that of the species studied in EFPC and WOC, would be introduced into Mitchell Branch.

Future bioindicator studies will involve placing a large enclosure containing hatchery-reared sunfish into lower Mitchell Branch above MIK 0.12 (Fig. 2-1). Fish will also be released outside the cage, and

Table 5.1. Biological indicators to be measured for sunfish (*Lepomis*) stocked in Mitchell Branch. These indicators represent both direct, indirect, and integrated effects of water quality, and were selected on the basis of previous studies of sunfish in both East Fork Poplar Creek below the Y-12 Plant and White Oak Creek near the Oak Ridge National Laboratory

Indicators of nutrition or indirect effects via the food chain	Indicators of water quality or direct effects of contaminants	Indicators which integrate direct and indirect effects
<ul style="list-style-type: none"> - serum triglycerides - total lipids - total body triglycerides - liver-somatic index (if liver enzymes low) 	<ul style="list-style-type: none"> - liver detoxification enzymes - SGOT - serum protein - phospholipid ratios - liver-somatic index (if liver enzymes high) - selected gill and liver histopathological parameters 	<ul style="list-style-type: none"> - growth - RNA/DNA ratio - total lipids - condition factor - liver-somatic index - visceral-somatic index - selected liver and spleen histopathological parameters

will be identified by individually numbered tags. At 3- to 5-wk intervals, subsamples of fish will be removed from inside and outside of the cage. These fish will be analyzed for the same suite of indicators selected in previous studies (Table 5-1). Differences in responses between caged (starved) and uncaged (well-fed) fish will be used to evaluate the relative importance of direct (water-borne) vs. indirect (food-chain) effects of stress on fish.

6. INSTREAM ECOLOGICAL MONITORING

The objectives of the instream ecological monitoring task (Task 4 of the BMAP as described in Loar et al. 1986b) are to (1) characterize spatial and temporal patterns in the distribution and abundance of the benthic macroinvertebrate and fish populations and (2) document the effects of new pollution abatement facilities on community structure and function. This task consists of two components: (1) benthic macroinvertebrate studies (Subtask 4a) and fish population studies (Subtask 4b); results to date of these studies are presented in Sects. 6.1 and 6.2, respectively.

6.1 BENTHIC MACROINVERTEBRATES

6.1.1 Introduction

Benthic macroinvertebrates are those organisms which are large enough to be seen without the aid of magnification and which live on or in the substrate of flowing and nonflowing bodies of water. Their limited mobility and relatively long life spans (a few weeks to more than a year) make them ideal for use in evaluating the ecological effects of effluents to streams (Platts et al. 1983). Thus, the composition and structure of the benthic community reflects the relatively recent past and can be considerably more informative than methods that rely solely on water quality analyses, but ignore the potential synergistic effects often associated with complex effluents.

The objectives of the initial phase of this study were to spatially and temporally characterize the benthic invertebrate community of Mitchell Branch. This information will be used as a baseline from which change can be followed during the monitoring phase of the program. The data will also be used to provide direction in future studies.

6.1.2 Materials and Methods

Benthic macroinvertebrates were sampled at approximately monthly intervals from August 1986 through July 1987 at six sites in Mitchell Branch (Fig. 6-1); the upstream most site (MIK 1.43) served as a

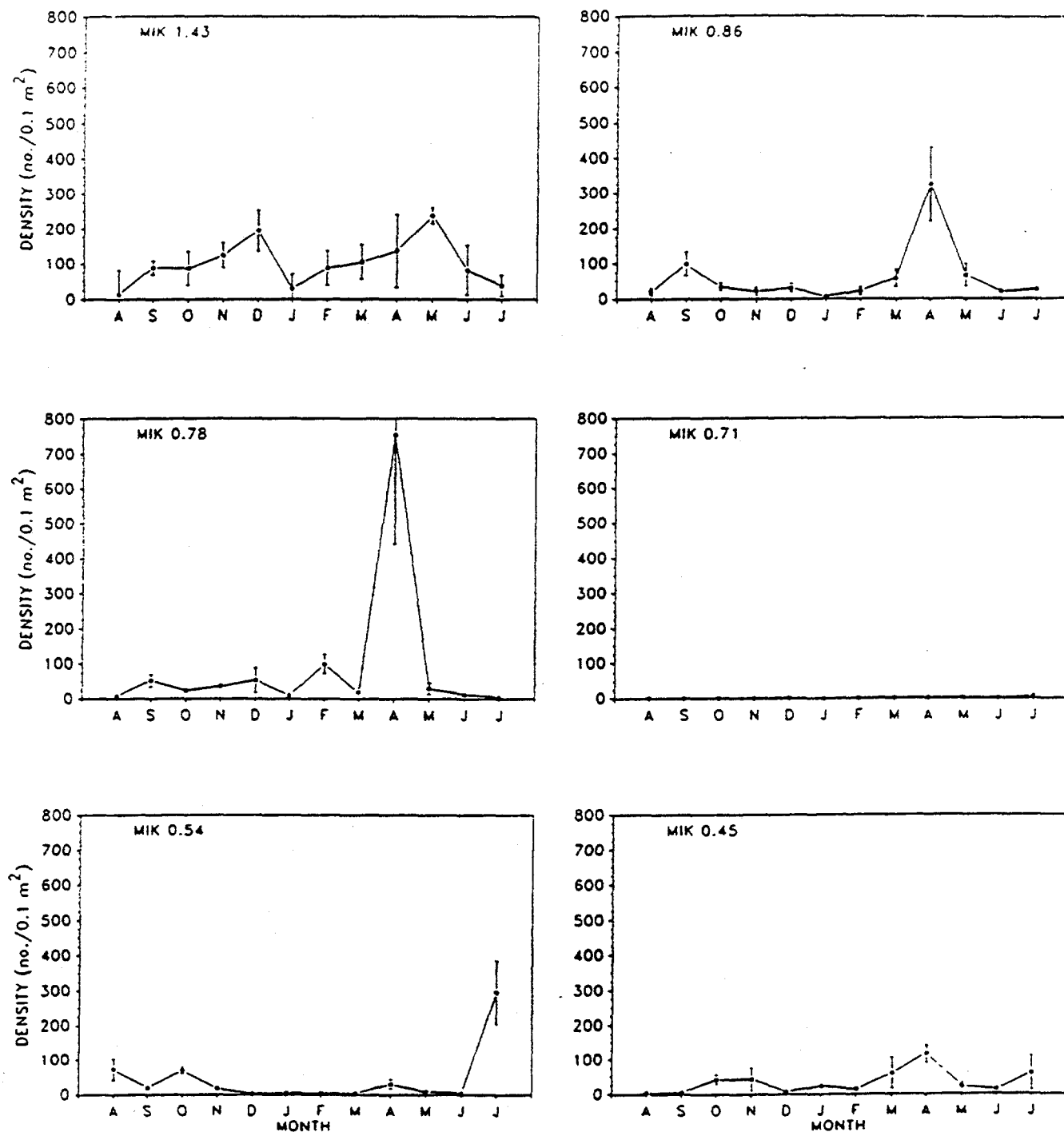


Fig. 6-1. Monthly mean density of benthic macroinvertebrates in Mitchell Branch, August 1986 through July 1987. Vertical Bars are ± 1 SE.

reference site. Quantitative samples were collected with a Surber bottom sampler (0.09 m^2 or 1 ft^2) fitted with a 363-micron-mesh collection net. Three randomly selected samples were collected from designated riffle areas at each site. To obtain additional information on the species richness at each site, qualitative samples were taken from riffle and non-riffle habitats (e.g., pools, leaf packs, detritus, snags, etc.) in the spring of 1987. The qualitative samples were collected with a D-frame aquatic dip net (mesh of 800×900 microns) and washed in the field in a handnet (363 microns) and white enamel pan in order to concentrate the organisms. All samples were placed in prelabeled glass jars and preserved in 80% ethanol; the ethanol was replaced with fresh ethanol within one week.

Various supplemental information was also recorded at the time of sampling. During the first nine months, water temperature and conductivity were measured at each site with a Col-Parmer Model R-1491-20 LCD temperature/conductivity meter. During the last three months temperature and conductivity, as well as dissolved oxygen, pH, and turbidity, were measured with an Horiba Model U-7 Water Quality Checker. Water depth, location within the riffle area (distance from permanent headstakes on the stream bank), relative stream velocity (very slow, slow, moderate or fast), and substrate type based on a modified Wentworth particle size scale (Loar et al. 1985b) were recorded for each sample.

All samples were washed in the laboratory using a standard no. 60 mesh (250-micron-mesh) sieve and placed in a large white tray. Organisms were removed from the debris with forceps and placed in labeled vials containing 70% ethanol. Organisms were identified to the lowest practical taxonomic level using a stereoscopic dissecting microscope. A blotted wet weight of all individuals in each taxon was determined to the nearest 0.01 mg on a Mettler analytical balance.

Chironomid larvae were identified from permanent slide mounts. Larvae were initially sorted into groups based on morphological similarities (i.e., body size, head capsule shape and coloration, abdominal setae, etc.), and then the head capsule and body of one or two larvae in each group were mounted on a microscope slide in CMC-10

mounting medium. The head capsules of larger larvae were first cleared in hot 10% potassium hydroxide solution for 10 min. and then mounted in CMC-10 mounting medium. After drying overnight, the mounted larvae were identified using a compound binocular microscope. Slides of mounted larvae were stored in slide boxes and retained for reference.

Individual taxa from a given site and sampling date were preserved in separate vials in 80% ethanol. A reference collection, for which the identification of each taxon has been verified, will be maintained at ORNL.

All data analyses were performed on transformed data [$\log_{10}(X+1)$] (Elliott 1977) using SAS (1985a,b). The Shannon-Wiener index (H') was used to calculate the taxonomic diversity of benthic macroinvertebrates at each site (Pielou 1977):

$$H' = - \sum P_j \log_2 P_j$$

where P_j is the proportion of the benthic invertebrate community made up by species j . Mean values for density, biomass, number of taxa (species richness), and diversity were compared separately with a one-way analysis of variance on month-specific data with site as the main effect. Significant differences ($\alpha = 0.05$) were then separated with a Tukey studentized range test.

Annual production of the benthic macroinvertebrates at each site was estimated indirectly by multiplying the mean annual biomass of each taxon by its respective production/biomass (P/B) ratio. P/B ratios were obtained from (1) values published for specific taxa in the upper Southeast region of the United States, or (2) a theoretical cohort P/B of 5.0 and then corrected for the cohort production interval (CPI = the length of aquatic life during which a taxon is present and growing) (Benke 1979; Waters 1977; Waters 1979). The CPIs were either obtained from published values, published life history information, or from the author's unpublished data. The P/Bs used in this report are listed in Appendix C, Table C-1.

6.1.3 Results

6.1.3.1 Taxonomic Composition

A total of 134 taxa were identified in quantitative samples collected from Mitchell Branch from August 1986 through July 1987 (Table 6-1). Of these taxa, 119 were insects representing 10 orders. The group containing the greatest number of taxa was the dipterans (true flies), with 66 taxa; 51 of these taxa were chironomids (nonbiting midges). The other orders of insects collected included Colembolla (spring tails), Coleoptera (beetles), Ephemeroptera (mayflies), Hemiptera (true bugs), Lepidoptera (butterflies and moths), Megaloptera (alderflies, dobsonflies, fishflies, and hellgramites), Odonata (damselflies and dragonflies), Plecoptera (stoneflies), and Trichoptera (caddisflies). The most taxonomically rich insect orders (other than dipterans) were the Trichoptera, Odonata, Coleoptera, Ephemeroptera, and Plecoptera which, were represented by 12, 11, 9, 7, and 7 taxa, respectively. The non-insect taxa collected included Amphipoda (sideswimmers), Decapoda (crayfish), Isopoda (aquatic sow bugs), Hydracarina (water mites), Nematoda (roundworms), Oligochaeta (worms), Turbellaria (flatworms/planarians), Gastropoda (snails), and Pelecypoda (clams, mussels); each of these groups was represented by one or two taxa. In addition to the taxa collected during routine quantitative sampling, 12 additional taxa were collected qualitatively in March 1986, (Table 6-1).

6.1.3.2 Density and Biomass

Mean annual density and biomass of the benthic invertebrates collected at each site in Mitchell Branch are presented in Table 6-2. Average density was significantly greater at MIK 1.43 (101.1 individuals/0.1 m²) than at all other sites (Tables 6-2 and 6-3). The lowest average abundance (2.0 organisms/0.1 m²) was found at MIK 0.71. Relative to the sites upstream of MIK 0.71, densities at MIK 0.54 and MIK 0.45 remained low but were significantly greater than at MIK 0.71.

The highest mean biomass occurred at MIK 0.86 (339.3 mg wet wt/0.1 m²) but was not significantly different from the biomass at

Table 6-1. Checklist of benthic macroinvertebrate taxa collected from Mitchell Branch, August 1986 through July 1987. An "X" denotes that the taxon was collected at least once in quantitative samples and a "Q" denotes that the taxon was collected in qualitative samples only

Taxon	Site					
	MIK 1.43	MIK 0.86	MIK 0.78	MIK 0.71	MIK 0.54	MIK 0.45
Turbellaria						
Planariidae	X		X	X		
Nematoda	X	X	X			
Oligochaeta	X	X	X	X	X	X
Crustacea						
Isopoda	X					
Asellidae						
Asellus	X	X				
Lirceus	X	X			X	
Amphipoda						
Gammaridae	X					
Crangonyx	X	X				
Decapoda	X	X				
Cambaridae						
Cambarus	X	Q				
Orconectes				X		X
Hydracarina					X	
Insecta						
Collembola	X	X	X	X	X	
Entomobryomorpha	X		X		X	X
Symphypleona			X			
Ephemeroptera						
Baetidae	X					
Baetis	X	X	X			
Caenidae						
Caenis				X		
Ephemerellidae	X					
Eurylophella	X					
Ephemeridae						
Ephemerella	X					
Leptophlebiidae	X					
Habrophlebiodes	X	X	Q			
Paraleptophlebia	Q					

Table 6-1. (Continued)

Taxon	Site					
	MIK 1.43	MIK 0.86	MIK 0.78	MIK 0.71	MIK 0.54	MIK 0.45
Odonata						X
Anisoptera						
Aeshnidae						
<i>Anax</i>						Q
<i>Boyeria</i>		X				
Cordulegastridae						
<i>Cordulegaster</i>	X	X	X			
<i>Cordulegaster</i>						
<i>maculata</i>	X					
Gomphidae	X					
<i>Ophiogomphus</i>						
<i>mainensis</i>	X					
<i>Stylogomphus</i>						
<i>albistylus</i>	X	X				
Libellulidae						
<i>Libellula</i>					Q	X
<i>Plathemis</i>	Q					
Zygoptera						
Calopterygidae						
<i>Calopteryx</i>	X	X	Q		X	
<i>Calopteryx</i>						
<i>maculata</i>	Q					
<i>Hetaerina</i>			X			
Coenagrionidae						
<i>Argia</i>		Q	Q		X	X
<i>Enallagma</i>	X				X	X
<i>Ischnura</i>						Q
Plecoptera	X	X				
Capniidae	X					
<i>Paracapnia</i>	X					X
Chloroperlidae						
<i>Sweltsa</i>	X	X				
Leuctridae	X					
<i>Leuctra</i>	X	X				
Leuctridae?	X					
Nemouridae						
<i>Amphinemura</i>						
<i>delosa</i>	X	X				
Perlidae	X					
<i>Eccoptura</i>						
<i>xanthenes</i>	X		Q			
Perlodidae	Q					

Table 6-1. (Continued)

Taxon	Site					
	MIK 1.43	MIK 0.86	MIK 0.78	MIK 0.71	MIK 0.54	MIK 0.45
Hemiptera	Q					
Gerridae	X				X	
Veliidae	X					
Megaloptera						
Corydalidae						
<i>Nigronia</i>						
<i>fasciatus</i>	X	X	X			
Sialidae						
<i>Sialis</i>	X	X				
Trichoptera	X					
Hydropsychidae						
<i>Cheumatopsyche</i>	X	X	X			
<i>Diplectrona</i>						
<i>modesta</i>	X		X			
Leptoceridae	X					
<i>Triacnoides</i>	X					
Limnephilidae						
<i>Goera</i>	X					
<i>Hydatophylax</i>	X				Q	
<i>Neophylax</i>	X					
<i>Pycnopsyche</i>	X					
Molannidae						
<i>Molanna</i>	X					
Philopotamidae						
<i>Chimarra</i>	X		X			
Psychomyiidae						
<i>Lype diversa</i>	X					
Lepidoptera	X					
Coleoptera						
Curculionidae	Q					
Dryopidae						
<i>Helichus</i>	X	X				
Dytiscidae						
<i>Hydroporus</i>	X					
Elmidae						
<i>Dubiraphia</i>	X					
<i>Optioservus</i>	X	X	X			
<i>Stenelmis</i>	X	X	X	X		
Hydrophilidae	X					
<i>Berosus</i>					X	

Table 6-1. (Continued)

Taxon	Site					
	MIK 1.43	MIK 0.86	MIK 0.78	MIK 0.71	MIK 0.54	MIK 0.45
Coleoptera (cont.)						
Ptilodactylidae						
Anchytarsus						
bicolor	X					
Staphylinidae			X	X		
Diptera	X					
Ceratopogonidae	X	X	X			
Chaoboridae						
Chaoborus				X	X	Q
Chironomidae						
Tanypodinae	X	X	X			
Larsia		X				
Larsia?	X	X	X			
Natarsia	X	X	X		X	X
Nilotanypus	X				X	
Nilotanypus?	X					
Paramerina	X					
Procladius					X	
Tanypus					X	
Thienemannimyia gp	X	X	X	X		X
Thienemannimyia gp?	X					
Trissopelopia		X	X			
Trissopelopia						
ogemawi	X		X			
Zavreliomyia	X	X	X			
Zavreliomyia?	X	X				
Diamesinae						
Diamesa		X	X		Q	
Diamesa?			Q			
Orthoclaadiinae	X	X	X			X
Chaetocladius		X	X		Q	
Chaetocladius?		Q		X		
Corynoneura	X	X	X			
Cricotopus						
bicinctus gp ^a		X	X	X	X	X
Cricotopus						
sylvestris gp ^a					X	X
Cricotopus						
tremulus gp ^a		X	X	X	X	X
Cricotopus/						
Orthocladus ^a	X	X	X	X	X	X
Heleniella	X	X				

Table 6-1. (Continued)

Taxon	Site					
	MIK 1.43	MIK 0.86	MIK 0.78	MIK 0.71	MIK 0.54	MIK 0.45
Orthocladiinae (cont.)						
<i>Heterotrissocladius</i>	X					
<i>Hydrobaenus</i>	X					
<i>Limmophyes</i>	X					
<i>Limmophyes?</i>	X					
<i>Nanocladius</i>	X					
<i>Parametriocnemus</i>	X	X	X		X	
<i>Parametriocnemus?</i>	X					
<i>Paraphaenocladius</i>			X			
<i>Psectrocladius</i>	X					
<i>Smittia</i>		X				
<i>Smittia?</i>					Q	
<i>Thienemanniella</i>	X					
<i>Tvetenia</i>	X					
Chironominae						
Chironomini	X					
<i>Chironomus</i>			X			
<i>Cryptochironomus</i>	X	X				X
<i>Cryptotendipes</i>	X					
<i>Dicrotendipes</i>	X	X	X	Q	X	
<i>Endochironomus</i>					X	
<i>Glyptotendipes</i>					X	
<i>Goeldichironomus</i>			X		X	X
<i>Microtendipes</i>	X		X			
<i>Paralauterborniella</i>	X					
<i>Paralauterborniella?</i>	Q					
<i>Paratendipes</i>	X		X	X	X	X
<i>Phaenopsectra</i>	X		X			
<i>Polypedilum</i>	X	X	X		X	X
<i>Stenochironomus</i>		X				
Tanytarsini	X	X	X			
<i>Microsectra</i>	X					
<i>Paratanytarsus</i>					X	
<i>Rheotanytarsus</i>	X					
<i>Rheotanytarsus?</i>	X		X			
<i>Stempellina</i>	X					
<i>Stempellinella</i>	X	X				
<i>Stempellinella?</i>	X					
<i>Tanytarsus</i>	X	X	X			X
Dolichopodidae	X					
Dixidae						
<i>Dixa</i>	X					

Table 6-1. (Continued)

Taxon	Site					
	MIK 1.43	MIK 0.86	MIK 0.78	MIK 0.71	MIK 0.54	MIK 0.45
Diptera (cont.)						
Empididae						
<i>Hemerodromia</i>	X	X	X	X	X	
Empididae?	X					
Muscidae						X
Sciomyzidae	Q					
Simuliidae	X	X	X	X	X	X
Stratiomyidae		X				
<i>Allognosta</i>	Q					
<i>Stratiomys</i>	Q					
Tabanidae	X		X			X
Tipulidae	X	X	X		X	
<i>Hexatoma</i>	X					
<i>Tipula</i>	X	X	X			
<i>Tipula abdominalis</i>	X		X			
Mollusca						
Gastropoda						
Ancylidae						
<i>Ferrissia</i>	X					
Lymnaeidae			X			
<i>Fossaria</i>						X
<i>Pseudosuccinea</i>						
<i>columella</i>	Q					
Physidae						
<i>Physella</i>			X	X		
Pelecypoda						
Sphaeriidae						
<i>Pisidium</i>	X					
<i>Sphaerium</i>	X					

^aBecause of the difficulty in reliably distinguishing the species groups within the genera *Cricotopus* and *Orthocladus*, they were combined into a *Cricotopus/Orthocladus* group for all data analyses.

Table 6-2. Mean density, biomass, richness, and diversity of benthic macroinvertebrates in Mitchell Branch, August 1986 through July 1987. Values in parentheses represent ± 1 SE

Site	Density (no./0.1m ²)	Biomass (mg wet wt/0.1m ²)	Richness (no. of taxa/sample)	Diversity (H')
MIK 1.43	101.1 (18.8)	218.1 (41.1)	20.2 (1.4)	3.52 (0.1)
MIK 0.86	60.7 (25.3)	339.3 (107.1)	6.3 (0.5)	1.69 (0.1)
MIK 0.78	91.9 (60.7)	231.0 (109.3)	5.6 (0.5)	1.52 (0.1)
MIK 0.71	2.0 (0.5)	2.8 (0.9)	1.1 (0.1)	0.24 (0.1)
MIK 0.54	44.3 (23.9)	34.2 (13.3)	2.9 (0.4)	0.91 (0.1)
MIK 0.45	34.5 (9.4)	75.3 (23.1)	2.4 (0.2)	0.60 (0.1)

Table 6-3. Statistical comparisons of mean benthic macroinvertebrate density, biomass, number of taxa per sample, and species diversity in Mitchell Branch, August 1986 through July 1987. Sites not joined by lines are significantly different ($\alpha = 0.05$), based on Tukey's studentized range test (HSD). N = 36 for each site

Parameter/Site					
<u>Density</u>					
MIK 1.43	MIK 0.86	MIK 0.78	MIK 0.45	MIK 0.54	MIK 0.71
<u>Biomass</u>					
MIK 1.43	MIK 0.86	MIK 0.78	MIK 0.45	MIK 0.54	MIK 0.71
<u>Number of Taxa</u>					
MIK 1.43	MIK 0.86	MIK 0.78	MIK 0.54	MIK 0.45	MIK 0.71
<u>Diversity</u>					
MIK 1.43	MIK 0.86	MIK 0.78	MIK 0.54	MIK 0.45	MIK 0.71

MIK 1.43 (218.1 mg wet wt/0.1 m²) (Tables 6-2 and 6-3). Biomass at these two sites was, however, significantly greater than at all other sites. Like density, the biomass at MIK 0.71 (2.8 mg wet wt/0.1 m²) was significantly lower than that at all other sites and remained low at MIK 0.54 and MIK 0.45 (34.2 and 75.3 mg wet wt/0.1 m², respectively).

Considerable temporal variation occurred in density and biomass both between and within sites, except at MIK 0.71 where both parameters remained low throughout the year (Figs. 6-1 and 6-2). Monthly densities ranged from a high of 754.2 organisms/0.1 m² at MIK 0.78 in April, to less than 1 individual/0.1 m² in September, November, and January at MIK 0.71. Mean monthly biomass ranged from a high of 1213.2 mg wet wt/0.1 m² at MIK 0.86 in December, to a low of 0.07 mg wet wt/0.1 m² at MIK 0.71 in September.

Except for MIK 0.78 and MIK 0.86, monthly trends in density and biomass within each site were generally similar (Figs. 6-1 and 6-2). Concurrent high values in density and biomass at MIK 0.78 and MIK 0.86 occurred only in April. The remaining peaks in biomass at these two sites were largely the result of a few large-bodied organisms within a single family (Tipulidae).

Temporal differences in density and biomass between the remaining four sites largely reflected differences in taxonomic composition. For example, the changes in total benthic density and biomass at MIK 0.45 and MIK 0.54 were due to changes in abundance of chironomids (primarily the *Cricotopus/Orthocladius* group) and oligochaetes. The temporal patterns observed at MIK 1.43 were considerably more complex. For example, an April peak in density was primarily the result of the combined densities of mayflies, stoneflies, and several chironomid taxa, and an April peak in biomass was primarily the result of the combined weights of dragonflies, crane flies, caddisflies, stoneflies, and oligochaetes. The peak in density at this site in December resulted largely from the combined densities of mayflies and several chironomid taxa, and the biomass peak resulted largely from the combined weights of dragonflies, crane flies, mayflies, and caddisflies.

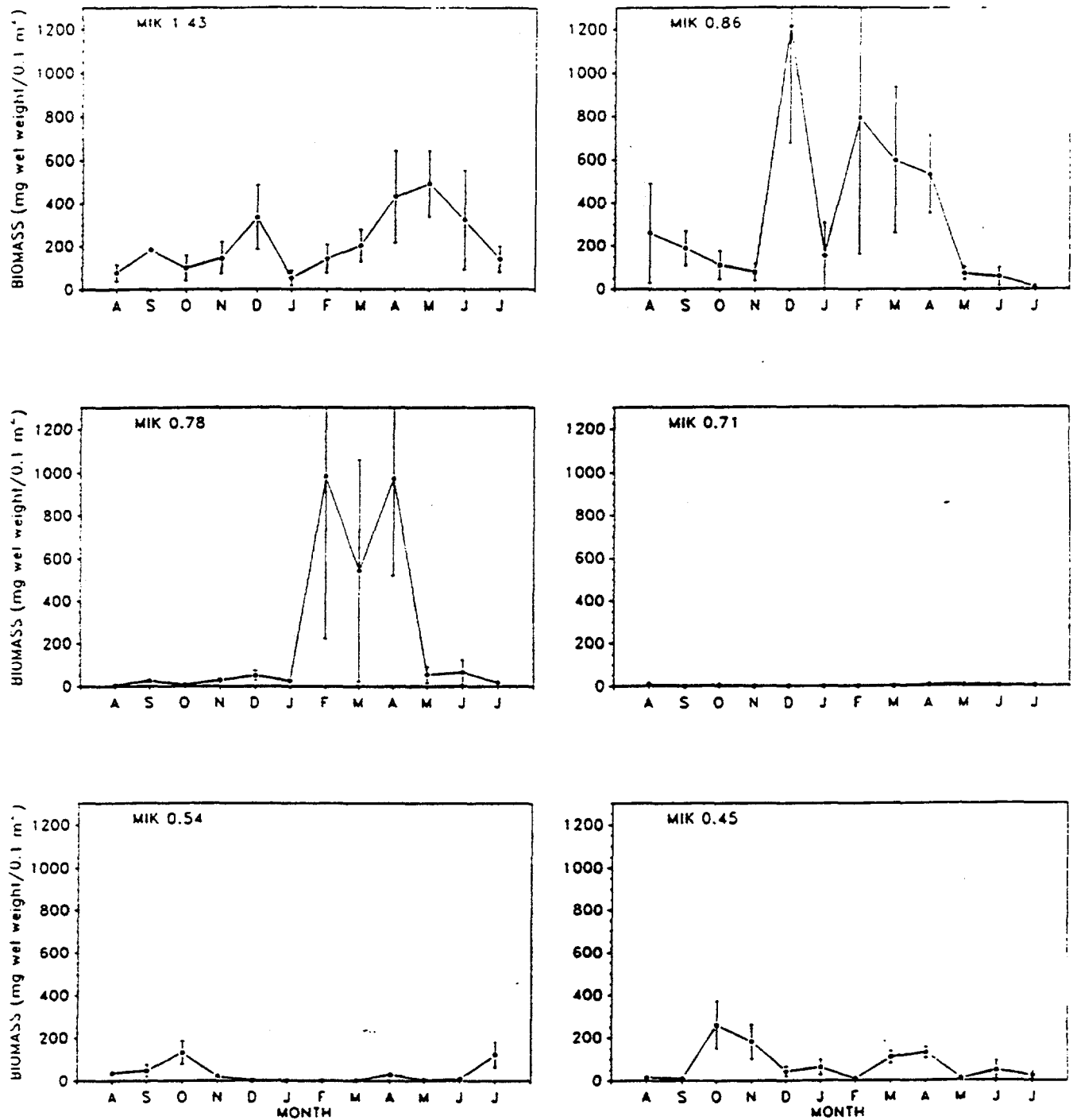


Fig. 6-2. Monthly mean biomass of benthic macroinvertebrates in Mitchell Branch, August 1986 through July 1987. Vertical bars are ± 1 SE.

6.1.3.3 Dominant Taxa

Most of the changes occurring in density and biomass of the benthic invertebrate community at each site could be attributed to two major groups: aquatic worms (Oligochaeta) and true flies (Diptera). At MIK 1.43, however, mayflies (Ephemeroptera) and stoneflies (Plecoptera) were sometimes major components of the community. Beetles (Coleoptera) were also occasionally major components of the communities at MIK 0.78, MIK 0.86, and MIK 1.43. The spatial trends of the dominant groups are presented in Fig. 6-3.

One of the most abundant groups of benthic invertebrates at all sites was the dipterans (Fig. 6-3). Numerically, the dipterans were comprised primarily of chironomids, which accounted for more than 33% of the total community density at all sites. Crane flies (Tipulidae) accounted for much of the biomass at some sites, particularly at MIK 0.78 (63%) and MIK 0.86 (56%); however, their densities were low and their occurrence was sporadic.

Oligochaetes were most likely represented by two or three families and several genera, but no identifications were made of this group. Oligochaetes were most abundant at MIK 0.45 where their mean annual density was 21.8 individuals/0.1 m² (63% of the total community density) (Fig. 6-3). Biomass of oligochaetes was highest at MIK 0.86 (77.0 mg wet wt/0.1 m²) and MIK 0.45 (66.0 mg wet wt/0.1 m²) where they accounted for 23% and 88%, respectively, of the total community biomass. The lowest density and biomass of oligochaetes occurred at MIK 0.71 (1.0 individuals/0.1 m² and 2.0 mg wet wt/0.1 m², respectively), but they still accounted for 48% and 71% of the total community density and biomass, respectively, at this site during the year.

The highest densities of mayflies and stoneflies occurred at MIK 1.43 where their mean annual densities were 13.7 and 10.0 individuals/0.1 m², respectively (Fig. 6-3). However, because of the diversity of organisms at this site, these two groups accounted for only 13% (mayflies) and 10% (stoneflies) of the total community density. The biomass of the mayflies and stoneflies at MIK 1.43 averaged 11.8 (5.4%) and 9.8 (4.5%) mg wet wt/0.1 m² respectively. Mayflies were absent at MIK 0.45 and MIK 0.54, and stoneflies were absent from MIK 0.54,

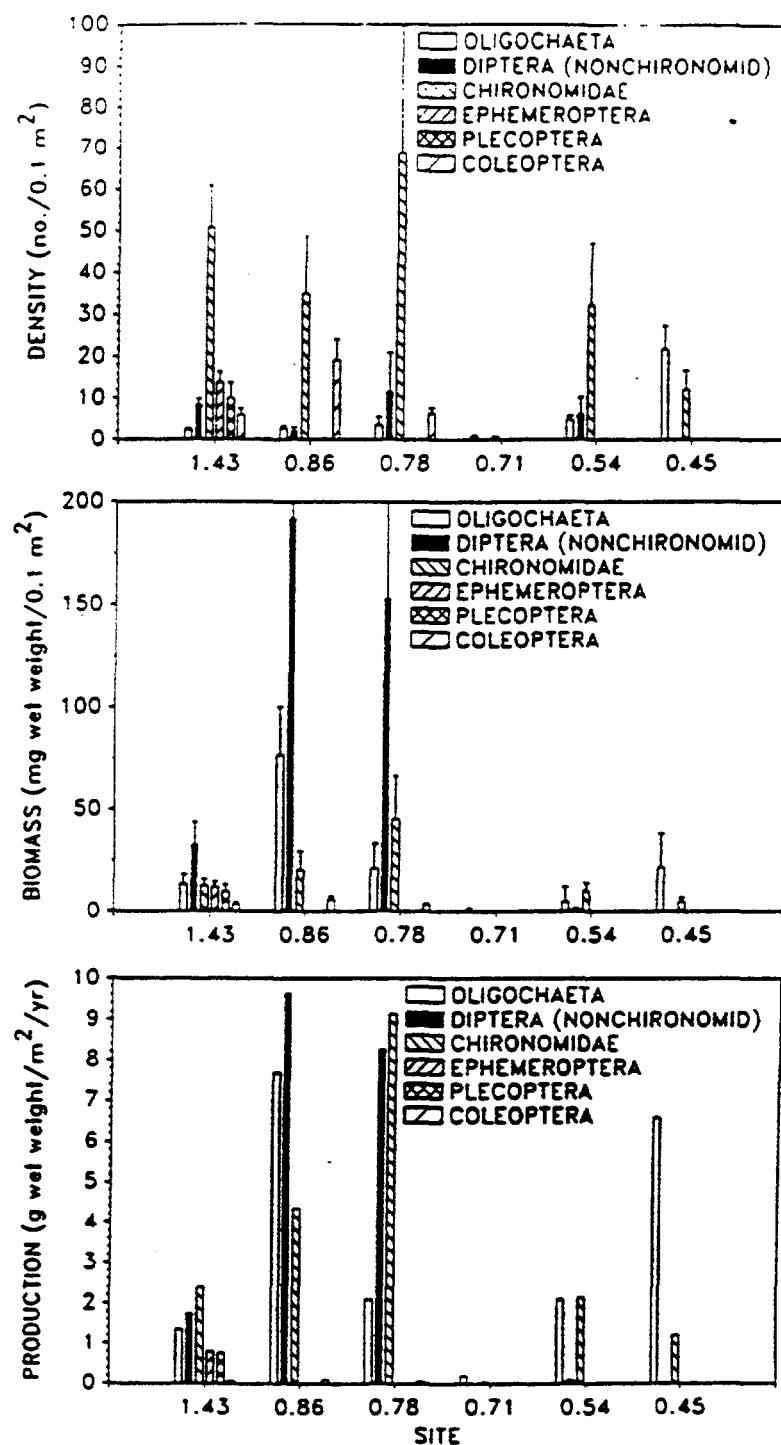


Fig. 6-3. Spatial trends in mean annual density, biomass, and production of Oligochaeta, Diptera (excluding Chironomidae), Chironomidae, Ephemeroptera, Plecoptera, and Coleoptera in Mitchell Branch, August 1986 through July 1987. Vertical bars are ± 1 SE.

MIK 0.71, and MIK 0.78. At the sites where they did occur (other than MIK 1.43), no more than two individuals were collected in any one month, and the mean annual density and biomass did not exceed 0.1 individuals/0.1 m² or 0.1 mg wet wt/0.1 m², respectively, for either group.

The greatest abundance of beetles occurred at MIK 0.86 where the average density was 19.3 individuals/0.1 m² (32%). They were also relatively abundant at MIK 1.43 and MIK 0.78 where their mean densities were 6.1 (6%) and 6.5 (7%) individuals/0.1 m², respectively. With the exception of MIK 0.71, their contribution to the total biomass did not exceed 2%; however, the mean biomass at MIK 0.71 was only 0.2 mg wet wt/0.1 m² (7.0%) versus, for example, 6.0 mg wet wt/0.1 m² at MIK 0.86.

6.1.3.4 Community Structure

Species Richness

The site with the highest total number of taxa (quantitative samples only) was MIK 1.43 (115 taxa) (Fig. 6-4). Considerably fewer taxa were collected at each site downstream of MIK 1.43. The number of taxa dropped to 53 and 48 at MIK 0.86 and MIK 0.78, respectively, and was lowest (15) at MIK 0.71. A small increase occurred farther downstream at MIK 0.54 (29) and MIK 0.45 (21).

The mean number of taxa collected per sample followed the same trend as the total number of taxa (Tables 6-2 and 6-3). A statistically significant reduction in the mean number of taxa occurred at all sites downstream of MIK 1.43 where a mean of 20.2 taxa/sample was observed. In general, there was a gradual but significant downstream decline in the mean number of taxa from MIK 0.86 (6.3 taxa/sample) to MIK 0.45 (2.4 taxa/sample). The exception to this gradual decline was MIK 0.71 which had significantly fewer taxa per sample (1.1) than any other site.

Seasonally, peaks in the mean number of taxa per sample generally occurred in the fall and spring (Fig. 6-5). These peaks were less distinct and variable at those sites with the lowest mean number of taxa per sample (i.e., MIK 0.45, MIK 0.54, and MIK 0.71).

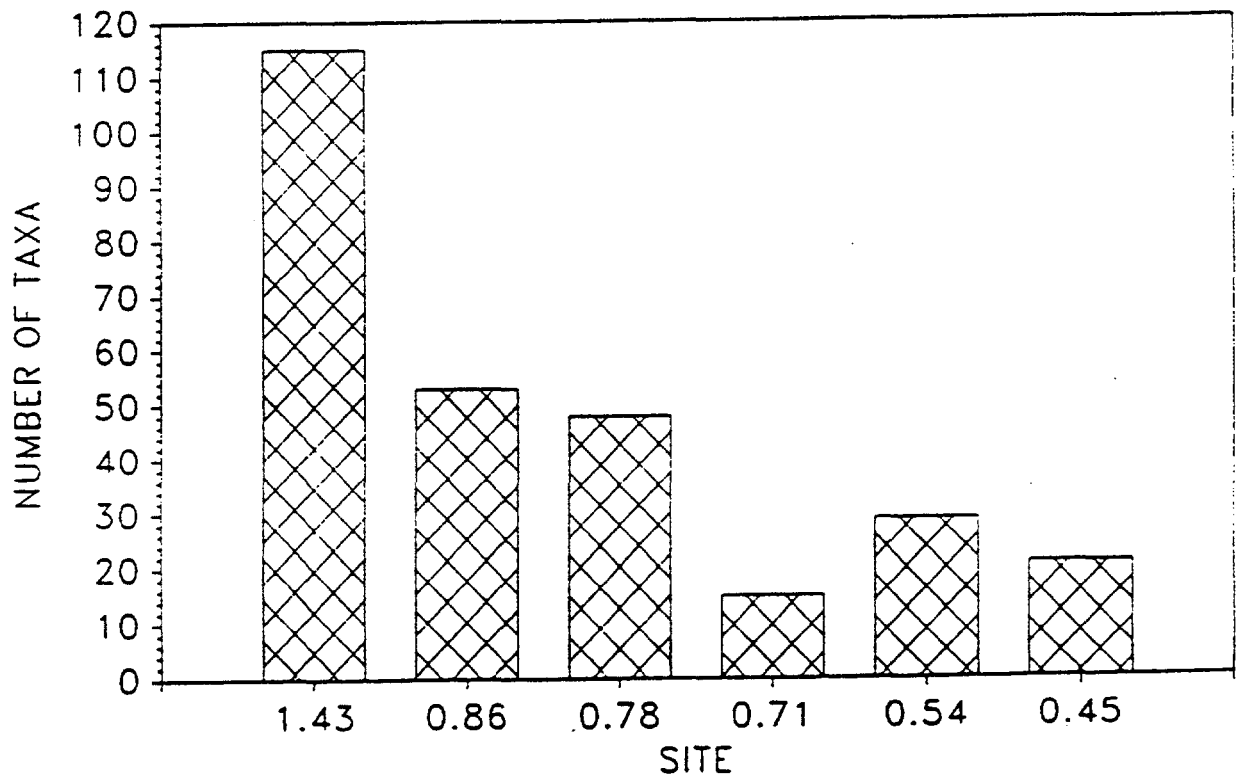


Fig. 6-4. Total number of taxa collected per site from Mitchell Branch, August 1986 through July 1987.

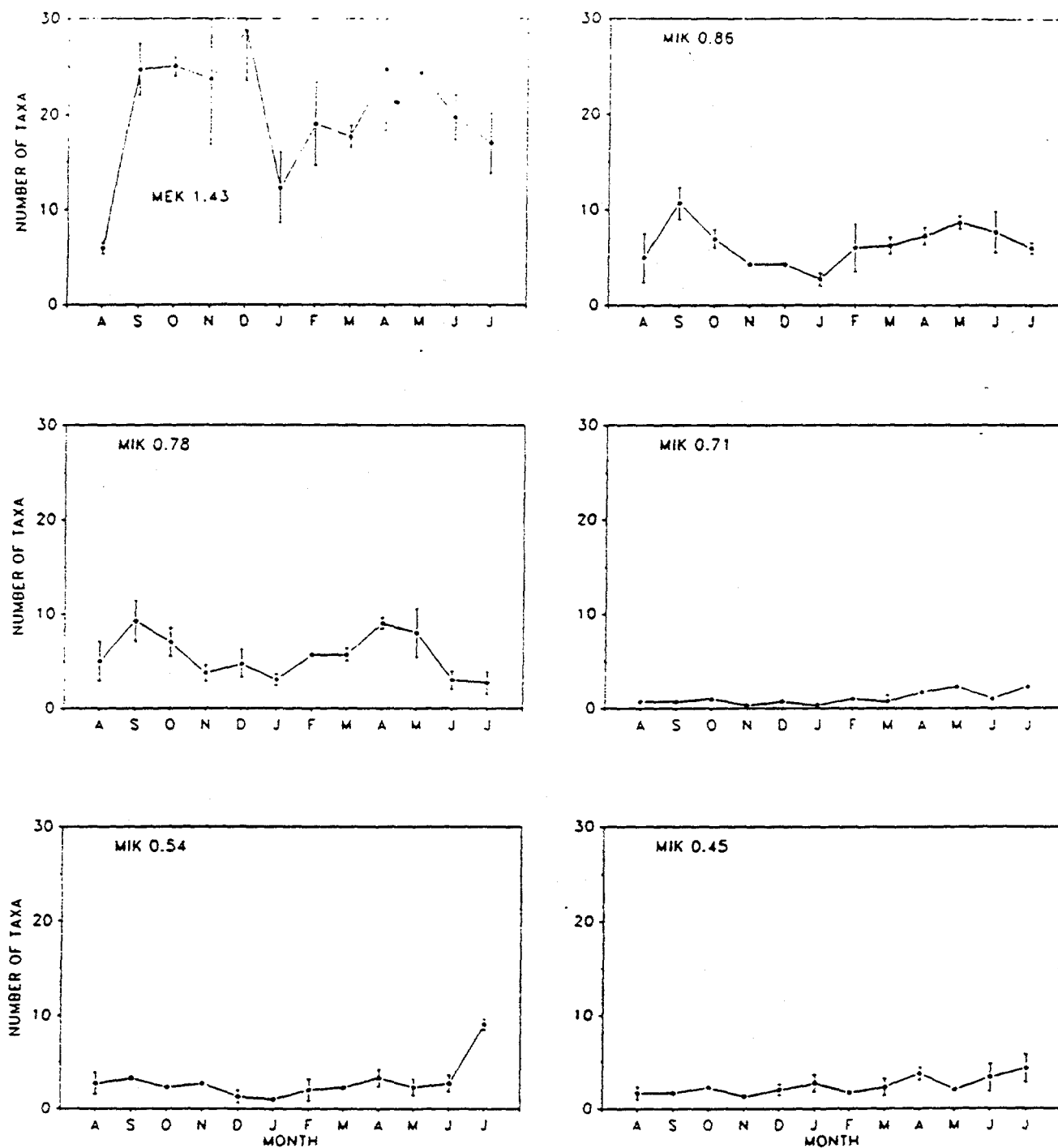


Fig. 6-5. Monthly mean number of benthic macroinvertebrate taxa per sample in Mitchell Branch, August 1986 through July 1987. Vertical bars are ± 1 SE.

Species Diversity

Species diversity (H') followed the same trends as species richness, with diversity decreasing significantly at all sites downstream of MIK 1.43 ($H' = 3.52$) (Tables 6-2 and 6-3). The lowest diversity was observed at MIK 0.71 ($H' = 0.24$), and was significantly lower than at all other sites.

Distinct seasonal trends in species diversity were not exhibited at any site (Fig. 6-6). Diversity at MIK 1.43 remained consistently high, falling below 3.0 in August only. Diversity at the other sites however, never exceeded 2.5 and rarely exceeded 2.0. Diversity at MIK 0.71 was 0.0 in all but four months, and at MIK 0.45 and MIK 0.54 diversity exceeded 1.0 and 1.5, respectively, only in July.

6.1.3.5 Secondary Production

Highest production of the benthic invertebrate community in Mitchell Branch occurred at MIK 0.86 (22.8 g wet wt/m²/yr) and lowest production was observed at MIK 0.71 (0.27 g wet wt/m²/yr) (Fig. 6-7). Intermediate levels of production were found at MIK 0.54, MIK 0.45, and MIK 1.43 (4.5, 8.0, and 10.9 g wet wt/m²/yr, respectively). More than 90% of the total annual community production of the benthic invertebrates at all sites except MIK 1.43 was attributable to two major taxa, Oligochaeta and Diptera, particularly the chironomids. At MIK 1.43, these groups accounted for only 51% of the total community production (Fig. 6-3). Although production of Ephemeroptera and Plecoptera was relatively low at MIK 1.43 (0.81 and 0.76 g wet wt/m²/year), these taxa still accounted for 7.4% and 7.0%, respectively, of the total community production; their contribution at the other sites was negligible. Production of Coleoptera was relatively low at all sites, exhibiting a maximum value of only 0.11 g wet wt/m²/yr at MIK 0.86.

6.1.4 Discussion

The benthic macroinvertebrate community of Mitchell Branch exhibited patterns indicative of increasing stress with increasing distance downstream. Maximum stress occurred in the midreaches, (around

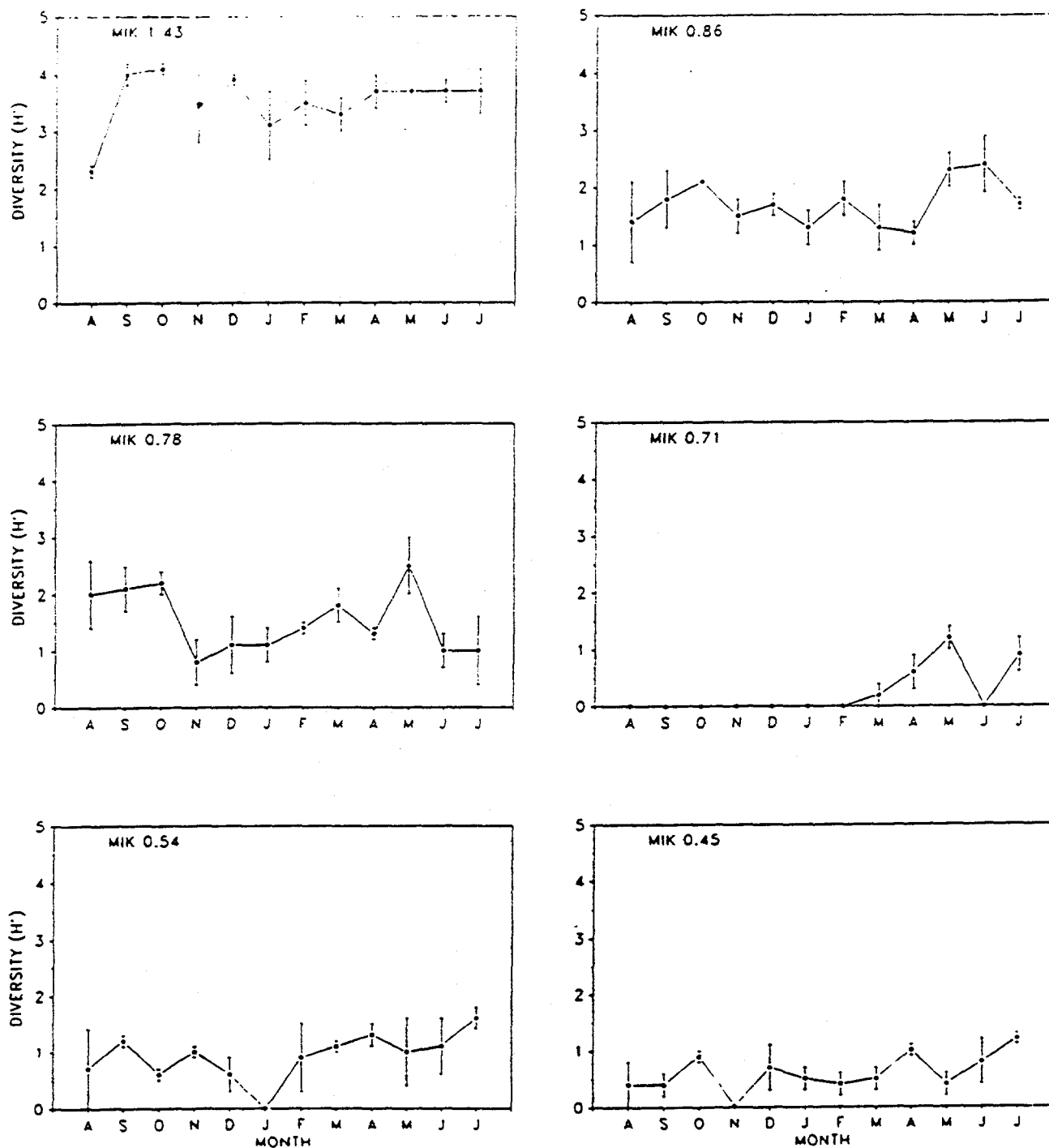


Fig. 6-6. Monthly mean species diversity of benthic macroinvertebrates in Mitchell Branch, August 1986 through July 1987. Vertical bars are ± 1 SE.

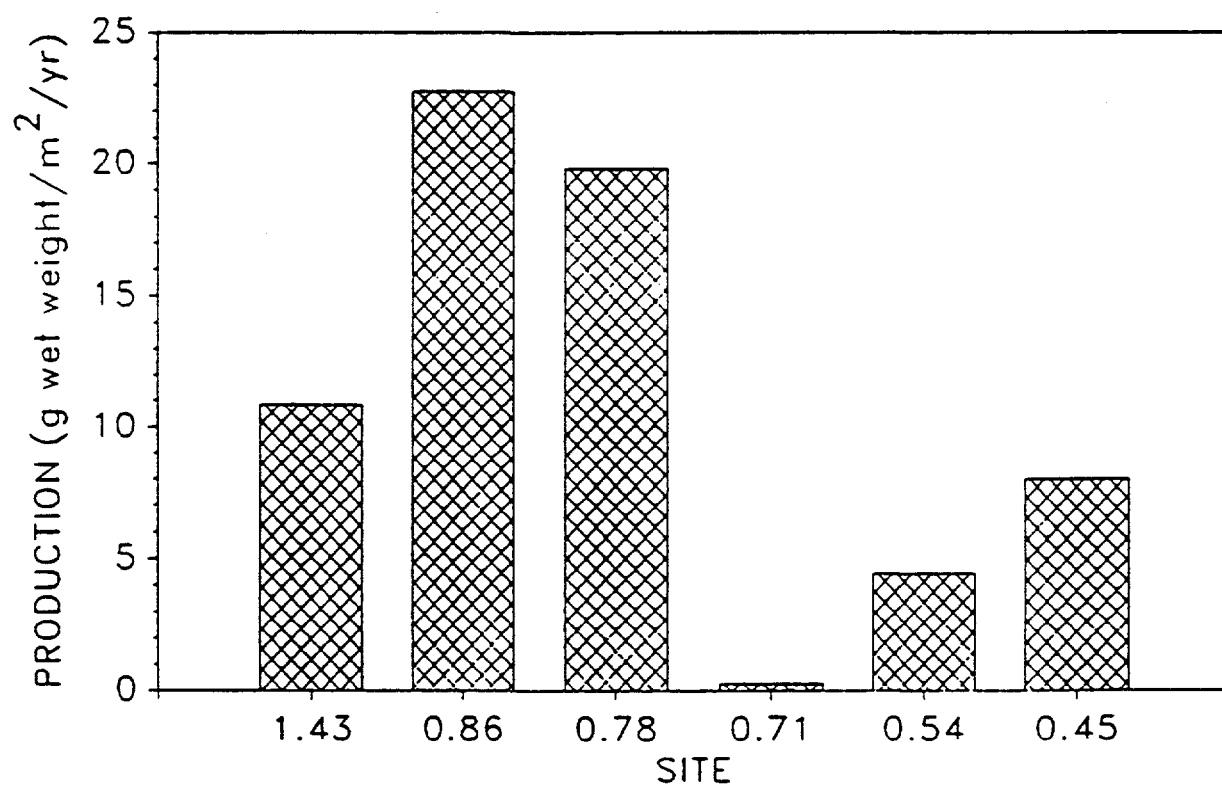


Fig. 6-7. Total community production of benthic macroinvertebrates in Mitchell Branch, August 1986 through October 1987.

MIK 0.71) with only minor recovery exhibited at the two most downstream sites. Density, species richness, and diversity were all significantly higher in the upstream reference site, MIK 1.43, and biomass was significantly greater at MIK 1.43 than all sites except MIK 0.86 (Table 6-3). This site was characterized by a very diverse and taxonomically rich community. Diversity values in clean bodies of water are typically equal to or greater than 3.0 (Platts et al. 1983). Diversity values at MIK 1.43 were greater than 3.0 in 11 of 12 months, and were similar to those found for benthic communities of relatively undisturbed reference streams used in other biological monitoring programs on the DOE Oak Ridge Reservation (ORR) (2.5 to 3.5) (Loar et al. 1987). Data on species richness from these same undisturbed reference streams indicate that the number of taxa collected per sample is typically 15 to 22, which is similar to the number of taxa collected per sample at MIK 1.43. Additionally, benthic densities invertebrate in reference streams on the ORR typically range from 50 to 400 individuals/0.1 m² (Loar et al. 1981a; Loar et al. 1986a; Loar et al. 1987) which is likewise similar to the densities observed at MIK 1.43. Biomass, on the other hand, is more variable and usually ranges between 200 and 800 mg wet wt/0.1 m². Thus, the benthic community at MIK 1.43 is similar to the benthic communities found in other relatively undisturbed streams on the ORR.

The benthic community at MIK 0.78 and MIK 0.86 exhibited characteristics of moderate stress (Tables 6-2 and 6-3). Diversity values typically ranged between 1 and 2 and were within the range generally considered indicative of moderate stress (i.e., 1 to <3; Platts et al. 1983). Diversity, richness, and density were all significantly lower at these sites than at MIK 1.43, but were significantly higher than at the other downstream sites. Biomass at MIK 0.86, however, was not significantly different from that at MIK 1.43. Although the mean annual biomass at MIK 0.78 was similar to that at MIK 1.43, biomass at MIK 0.78 was significantly lower due to considerable monthly variability.

The site in Mitchell Branch at which the benthic community exhibited the most stress was MIK 0.71. This site had the lowest

density, biomass, diversity, and richness of all six sites. Diversity values at this site were typically at the lower end of the range of values (0 to <1) considered indicative of heavy pollution (Platts et al. 1983).

The structure of the benthic communities at MIK 0.54 and MIK 0.45 was indicative of some recovery relative to MIK 0.71. Although diversity and the number of taxa per sample at these two sites were both significantly greater than at MIK 0.71, they were still significantly lower than at either MIK 1.43, MIK 0.86, or MIK 0.78. Density and biomass at both MIK 0.45 and MIK 0.54 were also significantly greater than at MIK 0.71, while differences in these parameters compared to MIK 0.78 and MIK 0.86 were only minor.

Taxonomic composition of the benthic invertebrate community in Mitchell Branch provided additional evidence of stress at those sites downstream of MIK 1.43. Chironomids were one of the most dominant groups at all sites; as the level of stress increased, chironomid richness decreased and the absolute and relative abundance of the *Cricotopus/Orthocladius* group increased. The benthic community at MIK 1.43 was very complex with other nonchironomid taxa contributing considerably to the community, including some groups which are generally intolerant of many kinds of pollution, such as the mayflies (Hubbard and Peters 1978) and stoneflies (Surdick and Gaufin 1978). All other sites in Mitchell Branch were characterized by considerably less complex communities that were dominated almost exclusively by chironomids and/or oligochaetes; these groups typically contain many pollution tolerant taxa (e.g. Hynes 1960). The taxonomic composition of the benthos community at MIK 0.54 may also have been influenced by discharges from the K-1407-B pond. For example, larvae of the dipteran *Chaoborus* are inhabitants of standing habitats (Cook 1981); their occurrence at MIK 0.54 was probably the result of drift into the site via the pond discharge. Other taxa may also have drifted into the stream from the pond and thereby artificially increased density, biomass, richness, and secondary production, and increased or decreased diversity.

The accuracy of secondary production estimates for benthic invertebrate communities depends upon a number of factors, including the

procedures and equipment used in collecting samples, the efficiency of processing samples, and the method(s) used for calculating production (Benke 1984; Waters 1979). Direct methods for calculating secondary production are the most accurate (Benke 1984) but require considerably more effort, which may not be feasible in studies such as a BMAP. Although methods that use production/biomass (P/B) ratios are less accurate than direct methods, they still provide a better estimate of the food potentially available to higher trophic levels than biomass alone. In addition, their accuracy can be enhanced when corrected for the cohort production interval (CPI) (Waters 1979). Such estimates also provide another means of comparing the "health" of benthic communities between sites and between years at a given site, and probably provide a better way of integrating changes occurring with other groups within an ecosystem (e.g. algae and fish).

Production of benthic invertebrates is controlled by factors such as food quality and quantity, temperature, habitat complexity, and biological complexity (Benke 1984). The relative importance of these factors can vary between streams and between sites within a stream. In Mitchell Branch, considerable differences in production occurred between sites. For example, production at MIK 0.86 was approximately 85 times greater than at MIK 0.71, and about two times greater than at MIK 1.43. The low production at MIK 0.71 is most likely a response to one or more toxicants, while the reasons for the difference in production between MIK 1.43 and MIK 0.86 are not currently known.

Because estimates of benthic invertebrate production in streams on the ORR and surrounding areas that are similar in size to Mitchell Branch are not yet available, the normal range of benthic invertebrate production in the Oak Ridge area is not yet known. However, some estimates of invertebrate community production exist for small, relatively undisturbed or minimally disturbed (e.g., runoff from cow pastures) streams do exist and are presented in Table 6-4. This table includes only those production estimates which were made on the entire benthic community from riffles only. These data suggest that benthic production at all sites in Mitchell Branch is considerably lower than in streams elsewhere. However, production of the benthic community at

Table 6-4. Published estimates of benthic invertebrate community production (g wet wt/m²/yr). Includes total community estimates from riffle habitat only

Stream	Production	Reference
Bear Brook	28.1	Fisher and Likens 1973
Caribou River	32.7	Kruger and Waters 1983
Blackhoof River	43.4	Kruger and Waters 1983
Ball Creek ^a	47.3	Huryn and Wallace 1987
Hinau Stream ^b S. Branch	50.4	Hopkins 1976
Rold Kilde ^a	62.3	Iversen 1988
North Branch Creek	132.4	Kruger and Waters 1983
Horokiwi Stream ^b Bush	147.78	Hopkins 1976
Bisballe Baek	150.0	Mortensen and Simonsen 1983
Hinau Stream ^b N. Branch	195.18	Hopkins 1976
Horokiwi Stream ^b Gyton	338.7	Hopkins 1976

^aEstimates were converted from ash free dry mass to wet mass assuming that dry mass is 17% of the wet mass and ash free dry mass is 90% of the dry mass (Waters 1977).

^bEstimates were converted from dry mass to wet mass assuming that dry mass is 17% of the wet mass (Waters 1977).

three sites in a second order southeastern blackwater stream (excluded from Table 6-4 because the estimate was for all habitats - riffles, pools, snags, etc.) ranged from 12 to 24 g wet wt/m²/year (Smock et al. 1985). Had the authors included estimates for riffles only, their estimates would likely have been less than, instead of similar to, the production estimates for the upper three sites in Mitchell Branch. Obviously, production varies considerably between streams and between sites within streams. Thus, estimates of secondary production for streams on and near the ORR will provide a better basis for future comparison with Mitchell Branch.

The benthic communities in the downstream sites of Mitchell Branch appear to be responding to several kinds of stress. Siltation, resulting from construction activities and clear cutting along much of the mid to upper reaches, may be a major perturbation at all sites downstream of MIK 1.43. The coarser substrates (e.g., gravels and cobbles) at MIK 0.71, MIK 0.78, and MIK 0.86 were typically covered with a light to heavy layer of silt while the substrate at MIK 0.45 and MIK 0.54 was almost exclusively fine gravel and silt. Silt can adversely affect organisms either directly (e.g. obstruction of food collection and/or respiration) or indirectly (e.g. reduction in food sources), and it typically provides conditions favorable to organisms such as chironomids and oligochaetes (Wiederholm 1984). This may partially explain the increase in the relative abundance of these two groups at all sites downstream of MIK 1.43.

In addition to silt, elevated levels of several other potential pollutants have been found in storm drains that enter Mitchell Branch upstream of MIK 0.86 and MIK 0.76 (Table 2-5; Scheib 1987). It is not known, however, whether these pollutants are present in Mitchell Branch at concentrations that are harmful to invertebrates.

Although siltation is evident in lower Mitchell Branch, the benthic communities at MIK 0.45, MIK 0.54, and MIK 0.71 appear to be responding primarily to frequent exposures to toxicants. This is suggested by the very low richness of taxa, the almost total absence of invertebrates that produce no more than two generations per year (e.g. mayflies, stoneflies, caddiflies), and the dominance of those invertebrates which

have the potential to produce three or more generation per year (Chironomidae). This hypothesis is consistent with the findings of the ambient toxicity studies (Sect. 3.2.4) and the fish population studies (Sect. 6.2.3.2). High levels of chlorine were found in storm drains (Sect. 2.2.3; Table 3-6) and in Mitchell Branch below SD170 (Fig. 3-1; Sect. 3.2.4), and may be a major source of toxicity (Mattice and Zittel 1976). Additionally, elevated levels of some metals and other compounds (e.g., various organics) were also found in lower Mitchell Branch (Table 2-5; Sect. 2.2.3; Table 3-3), many of which can be highly toxic to invertebrates (e.g., Hynes 1960 and Wiederholm 1984). The recovery of the benthic invertebrate community at MIK 0.45 and MIK 0.54 relative to MIK 0.71 indicates that the level and/or frequency of stress is reduced at these lower two sites.

Stream flow in Mitchell Branch is augmented (Sect. 2.1) and temperatures are elevated (Sect. 2.2.4) by effluent discharges. Both are perturbations that could influence the benthic communities at MIK 0.45, MIK 0.54, and MIK 0.71. Numerous studies have shown that alterations in flow and temperature from normal conditions can adversely affect benthic invertebrates in various ways (see reviews by Ward and Stanford 1979; Wiederholm 1984). Not only can extreme changes in these factors affect the benthos, but more subtle changes, such as reductions in their natural diel and seasonal fluctuations, can also result in adverse ecological impacts.

The ability of a benthic community to recover depends upon (1) duration, severity, and type of perturbation; (2) the physical and chemical characteristics of the receiving body of water; and (3) source of invertebrates for recolonization (Hynes 1960; Wiederholm 1984; Williams and Hynes 1976). One of the major sources of recolonization in streams is usually drift, although aerial and upstream migration may also contribute (Williams and Hynes 1976). As the biological stresses are reduced, recovery may be slowed somewhat due to the long-term nature of stress(es) to which the invertebrates in lower Mitchell Branch have been exposed. However, recovery will also be enhanced because the unimpacted headwaters of the stream will serve as a good source of organisms for recolonization.

6.1.5 Conclusions

The structure and composition of the benthic invertebrate communities in Mitchell Branch within the boundary of the ORGDP are indicative of adverse conditions. Maximum impact occurs within the midreaches of the stream just below SD170. Some minor recovery occurs downstream of this storm drain, thus indicating that discharges from SD170 may be the major source of stress.

The cause of the stress on the benthic communities at the impacted sites is most likely the result of several factors. The benthos between MIK 0.71 and MIK 0.45 appear to be responding primarily to exposures to toxicants (e.g., chlorine), although siltation and alterations in natural thermal and flow regimes are additional factors that could adversely impact the benthos. Siltation appears to be an important factor at MIK 0.78 and MIK 0.86, but periodic releases of other pollutants from storm drains just upstream of these sites may also be important. As stresses are reduced or eliminated, a steady recovery of the benthic fauna in Mitchell Branch should occur because unimpacted upstream areas are available to serve as a major source of organisms for recolonization.

6.1.6 Future Studies

The current program of monthly sampling of the benthic invertebrates will continue through July 1988, after which the sampling program will be reduced to a quarterly schedule. This will provide a comprehensive characterization of the benthos community in Mitchell Branch and an extensive data base from which future changes can be followed.

Data analyses in future reports will incorporate information obtained from reference sites used in other biological monitoring programs on the ORR. This information will help document the natural variability that occurs in the benthic populations and communities of other unimpacted streams in the Oak Ridge area.

6.2 FISHES

6.2.1 Introduction

Fish population and community studies can be used to assess the ecological effects of water quality and habitat degradation. These studies offer several advantages over other indicators of environmental quality (see Karr et al. 1986; Karr 1987) and are relevant to any evaluation of the biotic integrity of streams such as Mitchell Branch. Fish communities, for example, comprise several trophic levels, with species that are at or near the end of food chains. Consequently, they integrate the direct effects of water quality and habitat degradation on primary producers (periphyton) and consumers (benthic invertebrates) that are used for food. Because of these trophic interrelationships, the well-being of fish populations has often been used as an index of water quality (e.g., Weber 1973; Greeson et al. 1977; Karr et al. 1986). Moreover, statements about the condition of the fish community are better understood by the general public (Karr 1981).

The initial objectives of the instream fish monitoring task (Subtask 4b of the BMAP, as described in Loar et al. 1986) were to (1) characterize spatial and temporal patterns in distribution and abundance of fishes in Mitchell Branch, and (2) document any effects on fish community structure and function resulting from implementation of the ORGDP Water Pollution Control Program.

6.2.2 Methods

The fish community in Mitchell Branch was evaluated at five sites (Fig. 2-1 and Table 2-8) that reflect reaches potentially impacted by ORGDP effluents (Sect. 2.2.3); the upstream most site, MIK 1.43, served as a reference. Each site was sampled four times during the first year: September/October 1986, January 1987, May 1987, and October 1987. An additional reference site [Grassy Creek kilometer 2.4 (GCK 2.4)] to MIK 1.43 was sampled in December 1986, March 1987, and October 1987 (Fig. 2-2).

Fish populations were sampled using the three-pass removal method (Carle and Strub 1978). All stream sampling was conducted using a

Smith-Root Model 15A backpack electrofisher. Each unit has a self-contained, gasoline-powered generator capable of delivering up to 1200 volts of pulsed direct current. The pulse frequency and the output voltage can be varied, but generally a pulse frequency of 90 to 120 Hz and a voltage of less than 400 volts was used. The circular ring electrode at the end of the fiberglass anode pole was fitted with a nylon net (0.64-cm mesh) so that the operator could also collect stunned fish.

After a 0.64-cm-mesh block net was stretched across the upper and lower boundaries of the reach to restrict fish movement, a 2- to 3-person sampling team made three consecutive passes through the study reach in an upstream direction. If fish numbers captured during the first pass were low or zero, only one pass was made. Depending upon the turbidity of the water, all three passes could not always be made consecutively. Rather, fish were processed after each pass to allow sufficient time for the water to clear before another pass was started. Stunned fish were collected and placed by pass in wire mesh cages (0.64-cm diameter mesh) or buckets with small holes.

Fish were anesthetized with MS-222 (tricaine methanesulfonate), identified, measured to the nearest 0.1 cm (total length), and weighed using Pesola spring scales to the nearest 0.1 g (for fish less than 100 g) or gram (for fish greater than 100 g). At sites with high fish densities, individuals were tallied by 1-cm size class intervals and species. If 25 individuals of a species-size class were measured and weighed, additional members of that size class were only measured. Length-weight regressions (SAS 1985b) were later used to estimate missing weight data. Supplemental data recorded at the time of sampling included sex and reproductive state (if possible to determine), disposition (i.e., dead or kept for laboratory identification and reference collection), and presence of abnormalities (e.g., external parasites, skeletal deformities, etc.).

After processing fish from all passes, they were allowed to fully recover from the anesthetic and then returned to the stream within

the population reach. Any additional mortality occurring as a result of processing was noted at that time.

In addition to data taken on individual fish, measurements were made on physical and chemical conditions at the site during the sampling period. Prior to sampling, a Horiba Model U-7 Water Quality Checker was used to measure water quality including pH, temperature, dissolved oxygen, conductivity, and turbidity. Measurements were taken of stream width, depth, and length of the sampling reach after electro-shocking was completed (Table 6-5). Stream width was measured at transects taken at 5-m intervals in the sample reach, with depths recorded on the transect at three equally-spaced locations. These data were used to calculate the area sampled (Platts et al. 1983).

Data taken during population surveys were used to determine species richness, species density, and biomass. Species were identified using the taxonomic keys of Pflieger (1975), Eddy (1969), Etnier (1987), and Hubbs and Lagler (1964). Specimens that could not be determined in the field were preserved in 10% formalin, identified in the laboratory, and retained in a reference collection.

Calculation of population size was based on the multiple removal method (Seber and LeCren 1967), as modified to the maximum weighted likelihood analysis (Carle and Strub 1978). The population estimate thus obtained was divided by the stream area (mean width multiplied by total length of sampling reach) to determine density (number of individuals/m²).

Biomass values were calculated from length-weight data collected during each population survey. Using SAS programs (SAS 1985a,b) a mean weight was calculated for each species at each site for every collection period. These mean weights were multiplied by the species population density to determine biomass (g/m²).

Condition factors (K) were calculated for individual fish by site using the formula:

$$K = 100 (\text{weight}/\text{length}^3) ,$$

with weight in grams and total length in centimeters (Hile 1936). Fish without measured weights were not used in the calculation of mean

Table 6-5. Length, mean width, mean depth, and area of fish sampling sites on Mitchell Branch (MIK) and Grassy Creek (GCK), a reference stream, for each sampling date

Site (km)	Date (m)	Length (m)	Width (cm)	Depth (m ²)	Area
MIK 0.45	9/30/86	29	1.77	17.5	51
	1/21/87	30	2.04	19.6	60
	5/05/87	30	1.65	14.7	50
	10/01/87	32	1.71	14.1	55
MIK 0.54	9/30/86	43	1.43	10.7	62
	1/21/87	35	1.71	15.3	60
	5/05/87	40	1.27	8.8	51
	10/01/87	42	1.12	8.8	47
MIK 0.71	9/30/86	35	1.03	12.1	36
	1/21/87	48 ^a	1.03	16.5	50
	5/05/87	42	1.09	10.4	46
	10/01/87	56	0.88	10.7	49
MIK 0.78	10/01/86	25	1.14	16.9	28
	1/21/87	43 ^a	1.09	16.8	47
	5/06/87	36 ^b	0.94	9.0	34
	10/01/87	41	1.28	21.4	52
MIK 1.43	10/02/87	32	0.80	2.9	25
	1/28/87	31	1.1	5.3	34
	5/06/87	32	0.71	3.2	23
	10/02/87	32	0.55	2.1	18
GCK 2.4	NS ^c	-	-	-	-
	12/19/87	58	1.71	10.4	99
	3/25/87	59	1.63	9.6	96
	10/23/87	57	1.18	7.2	67

^aSite was lengthened following initial survey.

^bSite on this sampling date was divided temporarily into two sections by a 7-m concrete pipe to reduce siltation due to construction activities.

^cSite not sampled during this period.

condition factors. Comparisons of condition factors between sites were made using an ANOVA (SAS 1985b) on untransformed data because the condition factors exhibited homogeneity of the variance. If the ANOVA indicated significant differences in condition factors between groups, a Tukey studentized range test was performed to identify those groups that were significantly different (SAS 1985b).

6.2.3 Results and Discussion

Surveys of the sites made during the four periods indicated changes in the fish population densities and biomass. These changes, which are discussed below, reflected both extension of areas within Mitchell Branch that could not support fish populations and an overall decline in the total fish population of the stream.

6.2.3.1 Community Structure

The fish community of Mitchell Branch is a simple, three-member complex of tolerant species. Qualitative surveys of lower Mitchell Branch did not reveal any species not taken in the quantitative surveys. The blacknose dace, *Rhinichthys atratulus*, and the creek chub, *Semotilus atromaculatus*, are species common to the smaller first- and second-order streams in the Oak Ridge area (Ryon and Loar 1988). Redbreast sunfish, *Lepomis auritus*, also occur frequently in similar- and larger-sized streams in this area. A few large fish, perhaps adult redbreast, were observed in the lower reaches of Mitchell Branch below the population sites, and reproduction may occur in that region. All three species appear to be insensitive to many habitat and water quality stresses and would be considered tolerant species in an impact assessment methodology such as the Index of Biotic Integrity (Karr et al. 1986). Tropically, the blacknose dace is an omnivore, feeding on periphyton, as well as aquatic insects (Becker 1983). The redbreast sunfish and creek chub are both carnivores, feeding on fish and insects opportunistically (Becker 1983; Etnier 1987). In comparison with similar-sized, relatively undisturbed area streams, the fish species richness in Mitchell Branch is representative of headwater streams (Loar et al. 1987). In upper Grassy Creek at GCK 2.4, the striped shiner

(*Notropis chrysocephalus*) and white sucker (*Catostomus commersoni*), two omnivores, occur with blacknose dace and creek chub. In upper White Oak Creek at Kilometer 6.8 (WCK 6.8), the banded sculpin (*Cottus carolinae*), a carnivore, and the stoneroller (*Campostoma anomalum*), a herbivore, are found with blacknose dace and creek chub. These faunal differences may reflect temperature or water quality differences between the sites, although only the sculpin is considered to be an intolerant species (Karr et al. 1986).

6.2.3.2 Population Densities

The initial survey indicated that no fish were present at MIK 0.71; three subsequent surveys conducted during 1987 confirmed this observation (Table 6-6). The only fish taken at this site were collected in the May 1987 sample at the upper end of the sampling reach. These large individuals of creek chubs and redbreast sunfish may have been displaced from the next upstream section, MIK 0.78, due to construction activity near SD170 at MIK 0.76. The effluents entering Mitchell Branch from SD170 were toxic to fathead minnows in laboratory toxicity tests (Table 3-8). Episodic releases of chlorine (Sect. 2.2.3, Table 3-6) may prevent fish from establishing permanent residency in this section of Mitchell Branch. The greatest impact on the benthic invertebrate community also occurred at this site (Table 6-3).

The 1987 surveys also documented declines in fish populations at MIK 0.54 and MIK 0.45. In the initial survey conducted in October 1986, population densities of the three existing species (blacknose dace, creek chub, and redbreast sunfish) were similar to those found at the reference sites. During 1987, fish densities at these two sites approached or reached zero. The declining densities suggested a downstream extension of the stresses observed at MIK 0.71, or possibly the introduction of additional stresses due to discharges from K-1407-B holding pond, SD180, SD190, or seepage from the K-1407-C pond (Ashwood et al. 1986). The effluents from SD180 and SD190 enter Mitchell Branch within 50 m to 90 m of these sites. Elevated levels of chlorine and/or other pollutants were found in effluents from both drains and from the

Table 6-6. Fish densities (number fish/m²) in Mitchell Branch (MIK) and Grassy Creek (GCK), a reference stream, for September 1986 to October 1987

Species ^a	MIK 0.45	MIK 0.54	MIK 0.71	MIK 0.78	MIK 1.43 ^b	GCK 2.1
SEPTEMBER/OCTOBER 1986						
Blacknose dace	0.04	0.07	-	0.74	0.47	NS ^c
Creek chub	0.37	0.33	-	3.40	0.23	NS
Redbreast sunfish	0.53	0.11	-	0.81	-	NS
TOTAL	0.94	0.51	NF ^c	4.95	0.70	NS
JANUARY 1987						
Blacknose dace	0.07	-	-	0.79	0.06	0.6
Creek chub	0.10	-	-	2.66	-	0
Striped shiner	-	-	-	-	-	0.0
Redbreast sunfish	0.03	-	-	0.34	-	-
White sucker	-	-	-	-	-	0
TOTAL	0.20	NF	NF	3.79	0.06	1.1
MAY 1987						
Blacknose dace	0.06	0.04	-	0.12	0.30	0.6
Creek chub	0.06	-	0.13	1.00	0.09	0
Striped shiner	-	-	-	-	-	0.0
Redbreast sunfish	0.04	-	0.11	0.12	-	-
White sucker	-	-	-	-	-	0
TOTAL	0.16	0.04	0.24	1.24	0.39	0.6
OCTOBER 1987						
Blacknose dace	0.04	-	-	0.52	0.11	0.6
Creek chub	-	-	-	1.98	0.11	0
Redbreast sunfish	-	-	-	0.10	-	-
White sucker	-	-	-	-	-	0.0
TOTAL	0.04	NF	NF	2.60	0.22	1

^aSpecies are blacknose dace, *Rhinichthys atratulus*; creek chub, *Semotilus atromaculatus*; striped shiner, *Notropis chrysocephalus*; redbreast sunfish, *Lepomis auritus*; and white sucker, *Catostomus commersoni*.

^bReference site on upper Mitchell Branch.

^cNF = no fish taken in sample.

^dNS = not sampled.

^eSampled on December 12, 1986.

^fSampled on March 25, 1987.

K-1407-B holding pond (Tables 2-5, 3-3, and 3-6). The effects decreased downstream, as MIK 0.45 always supported some fish and at higher levels than did MIK 0.54 (Table 6-6). A similar trend was observed in benthic invertebrate production between the two sites (Fig. 6-7).

One site, MIK 0.78, was identified in the initial survey as an area of high fish density, perhaps serving as a refuge from toxicity downstream. The densities at this site during 1987 declined for all three species, particularly for the redbreast sunfish (Table 6-6). Although effluent discharges into the lower portion of this site may have been responsible for the decline, significant construction activity also occurred near the site during the latter two sampling periods. During one period in the spring and summer of 1987, the site was temporarily divided by a concrete pipe positioned in the stream to reduce amounts of silt-laden runoff from the construction area from entering the stream. The impacts of this activity could have dispersed the resident fish population from the site. However, the decline in abundance at MIK 0.78 was most likely due to effluents from SD170, which enter Mitchell Branch in the middle of the population sampling reach. No fish were collected at this site below this effluent discharge in the May and October 1987 surveys.

Sites MIK 1.43 and GCK 2.4 represent relatively unaffected reference areas. MIK 1.43 is located outside the ORGDP boundary, and although densities measured during 1987 showed a decline, this reduction is most certainly related to the drought (Sect. 2.1). The Grassy Creek reference site (GCK 2.4) has more stable flows and is more similar in size to the downstream sites in Mitchell Branch. The fish densities at GCK 2.4 during 1987 fluctuated, but remained at levels higher than most sites in Mitchell Branch.

6.2.3.3 Population Biomass

Fish biomass values (g/m^2 wet wt) paralleled the trends observed in population density (Table 6-7). Biomass at MIK 0.45 and MIK 0.54 declined in the same pattern as density and was less than the biomass at GCK 2.4. Like density, biomass at MIK 0.78 declined more than 50%

Table 6-7. Fish biomass (g/m² wet wt) in Mitchell Branch (MIK) and Grassy Cr (GCK) a reference stream, during September 1986 to October 1987

Species ^a	MIK 0.45	MIK 0.54	MIK 0.71	MIK 0.78	MIK 1.43 ^b	GCK 2.1
SEPTEMBER/OCTOBER 1986						
Blacknose dace	0.15	0.21	-	0.49	0.70	NS ^d
Creek chub	3.14	1.60	-	9.30	1.04	NS
Redbreast sunfish	2.51	0.62	-	2.96	-	NS
TOTAL	5.80	2.43	NF ^c	12.75	1.74	NS
JANUARY 1987						
Blacknose dace	0.27	-	-	2.6	0.12 ^e	0.1
Creek chub	0.83	-	-	12.0	-	1.1
Striped shiner	-	-	-	-	-	0.1
Redbreast sunfish	0.11	-	-	1.9	-	-
White sucker	-	-	-	-	-	0.1
TOTAL	1.21	NF	NF	16.5	0.12	2.3
MAY 1987						
Blacknose dace	0.25	0.20	-	0.12	0.75	0.1
Creek chub	0.92	-	1.6	4.6	0.50	0.8
Striped shiner	-	-	-	-	-	0.2
Redbreast sunfish	0.27	-	1.1	0.92	-	-
White sucker	-	-	-	-	-	0.3
TOTAL	1.44	0.20	2.7	5.64	1.25	2.3
OCTOBER 1987						
Blacknose dace	0.21	-	-	0.42	0.24	0.1
Creek chub	-	-	-	6.7	1.3	1.3
Redbreast sunfish	-	-	-	0.42	-	-
White sucker	-	-	-	-	-	1.1
TOTAL	0.21	NF	NF	7.54	1.54	3.1

^aSpecies are blacknose dace, *Rhinichthys atratulus*; creek chub, *Semotilus atromaculatus*; striped shiner, *Notropis chrysocephalus*; redbreast sunfish, *Lepomis auritus*; and white sucker, *Catostomus commersoni*.

^bReference site on upper Mitchell Branch.

^cNF = no fish taken in sample.

^dNS = not sampled.

^eEstimated population biomass based on the one specimen collected.

^fSampled on December 12, 1986.

between January and May 1987, again indicating that adverse impacts affecting the fish population occurred in Mitchell Branch at this time. When compared with the reference site on Grassy Creek (GCK 2.4) and another similar sized reference site on upper White Oak Creek (WCK 6.8; Loar et al. 1987), biomass at MIK 0.78 was high.

6.2.3.4 Condition Factors

Comparisons of condition factors (K) should provide information on the relative well-being of the fish, because those with more weight per length have a higher condition factor (Everhart et al. 1975). Data were available for a statistical comparison of condition factors for blacknose dace, creek chubs, and redbreast sunfish between sites during March/May 1987 and October 1987. This comparison revealed no significant differences between sites in either period ($\alpha = 0.05$). The similarity in condition of fish in Mitchell Branch and Grassy Creek, suggested that individual fish surviving stressful conditions in Mitchell Branch generally remained in good health. However, caution must be exercised in the interpretation of condition factors, because they may be relatively insensitive to environmental conditions or nutritional status (Loar et al. 1985b).

6.2.4 Conclusions

The data collected on the fish populations in Mitchell Branch during the 1986-1987 sampling period indicated a significant adverse impact. Species richness, density, and biomass of fish in October 1986 demonstrated that Mitchell Branch supported the expected, limited fauna, with some impacts related to ORGDP operations. When the same parameters were evaluated over the next year, a steady decline in the robustness of the fish community was observed. The absence of fish below SD170, which enters Mitchell Branch at MIK 0.76, strongly suggests that toxic effluents are entering the stream at this site. Toxic insults may have also been added through discharges from K-1407-B, SD180 and SD190. Although residual chlorine is the most likely toxicant, other pollutants may also be partially responsible for the observed effects on the fish populations.

6.2.5 Future Studies

The original plan for assessing the fish community in Mitchell Branch involved quarterly sampling of the fish populations for the first year at a minimum (Loar et al. 1986b). Because the decline in density and biomass indicates stress at the community level, the quarterly sampling regime at the sites on Mitchell Branch and Grassy Creek will continue for at least another year. Because 1987 data indicated an adverse impact from MIK 0.76 downstream to MIK 0.45, and attempt will be made to quantitatively assess impacts in the impounded area of lower Mitchell Branch. This quantitative assessment may involve a three-pass removal estimate using a fixed period of sampling effort as a comparative measure.

Plans to assess impacts at the individual level, by examining fecundity or feeding patterns, will be implemented next year only if population densities and biomass stabilize. Estimates of production of the three fish species will be calculated using the method of Garman and Waters (1983). Because immigration of individuals into Mitchell Branch from Poplar Creek is possible (although difficult due to a downstream weir), further qualitative sampling will be conducted to assess immigration. Also, the capability for reproduction by resident adult redbreast sunfish in lower Mitchell Branch will be evaluated by sampling in the spring/summer to examine the reproductive status of the population or the presence of larval fish in Mitchell Branch.

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APPENDIX A

QUALITY CONTROL ANALYSES

APPENDIX A QUALITY CONTROL ANALYSES

The number of samples analyzed in the BMAP for Mitchell Branch was too small to evaluate statistically. However, results for blind duplicate analyses are shown in Table A-1 and were typical of those observed in the Y-12 Plant BMAP (Loar 1988). Analyses of reference tissues indicated good recovery and quantification of metals, and recovery of known amounts of PCBs added to uncontaminated fish or clam tissues was excellent (Table A-1). Because the analyses of the Mitchell Branch samples were conducted at the same time as the analyses of the much larger group of Y-12 BMAP samples from EFPC, quality assurance data on the latter group of samples are presented to provide a more statistically oriented evaluation. Blind duplicate samples (15 pairs) analyzed for mercury showed a relatively low degree of variation, with a mean coefficient of variation (CV) between sample pairs of 11% and a mean standard deviation (SD) of 0.05 ppm. The mean difference between duplicate samples was 0.04 ppm. The multiple analyses of EPA reference fish for mercury ($n = 21$) agreed very well with the expected value, averaging 2.55 ± 0.09 ppm (mean \pm SD) versus an expected value of 2.52 ppm for an average recovery of $101 \pm 4\%$. Split duplicate fish samples analyzed for mercury by the ORNL Analytical Chemistry Division (ACD) and the EPA Environmental Services Laboratory in Athens, GA were in good agreement. Fish analyzed by ORNL averaged 1.10 ppm mercury, while those analyzed by the EPA lab averaged 1.17 ppm. The mean difference between individual samples analyzed by EPA and ORNL (0.06 ± 0.11 ppm; CV = 10%) was not significantly different from zero ($\alpha = 0.05$) and both the CV and SD were similar to those for duplicate analyses within the ACD/ORNL Laboratory. Mercury levels in sunfish from the uncontaminated reference site (Hinds Creek) were typical of background levels in stream fish (TVA 1985), averaging 0.09 ± 0.03 ppm (mean \pm SD, $n = 11$).

The results of PCB analyses of 12 pairs of blind duplicate fish samples were much more variable than the mercury analyses, as is generally the case. The mean difference between duplicates was

Table A-1. Results of quality assurance analyses of Asiatic clam (*Corbicula fluminea*) and redbreast sunfish (*Lepomis auritus*) samples from Mitchell Branch, 1987.
NA = not analyzed

A. Blind duplicate

Sample ID	<u>Concentration (ppm, wet wt)</u>	
	Hg	PCBs
7025	0.38	0.46
5207	0.60	0.29
7055	0.27	NA
5507	0.23	NA

B. National Bureau of Standards/EPA reference tissue

Metal	Sample type ^a	<u>Concentration (ppm, dry wt)</u>	
		Expected	Found
As	O	13.4	11.9
Cd	O	3.5	3.2
Cu	F	2.2	2.3
Cr	F	0.58	0.49
Hg	F	2.52	2.57
Ni	O	1.0	1.0
Pb	O	0.48	0.46
Se	O	2.1	2.1
Zn	F	44.0	43.0

C. Spiked sample recovery

Sample type	<u>% recovery</u>	
	PCB-1254	PCB-1260
Clam	85	Not Added
Fish	100	106
Fish	96	113

^aO = Freeze-dried oyster tissue, National Bureau of Standards No. 1566.

F = Freeze-dried fish tissue, EPA trace metals on fish, U.S. Environmental Protection Agency, Cincinnati, Ohio.

0.27 ± 0.25 ppm (CV = 30%). The variabilities in the measurement of PCB-1254 and PCB-1260 were similar, with mean differences between duplicates of 0.16 ± 0.13 (CV = 24%) and 0.11 ± 0.14 (CV = 38%), respectively. Samples of uncontaminated fish [bluegill (*Lepomis macrochirus*), redbreast sunfish (*L. auritus*), and carp (*Cyprinus carpio*)] were spiked with 1 ppm PCB-1254 and 1 ppm PCB-1260 and analyzed along with fish samples. Recoveries of nine samples averaged $94 \pm 6\%$ for total PCBs, and $91 \pm 9\%$ and $99 \pm 8\%$ for PCB-1254 and PCB-1260, respectively. Samples of PCB-contaminated carp and channel catfish (*Ictalurus punctatus*) were homogenized and split for analysis by the ACD/ORNL Laboratory and the EPA Environmental Services Laboratory in Athens, GA. The mean concentrations of total PCBs, PCB-1254, and PCB-1260 were not significantly different between the two laboratories ($\alpha = 0.05$, $n = 5$), but the results obtained by the ORNL lab were somewhat higher, averaging 1.35 ± 0.88 vs 0.94 ± 0.34 , 0.52 ± 0.55 vs 0.40 ± 0.19 , and 0.83 ± 0.66 vs 0.51 ± 0.28 ppm for total PCBs, PCB-1254 and PCB-1260, respectively. The variability between duplicate samples analyzed at the ORNL and EPA laboratories was similar to the variability between duplicates analyzed at ORNL (SD = 0.39, 0.21, and 0.24, and CV = 30, 61, and 28%) for total PCBs, PCB-1254 and PCB-1260, respectively. Samples of 12 redbreast and bluegill sunfish from Hinds Creek were used as analytical controls; these exhibited very low levels of total PCBs, averaging 0.04 ± 0.04 ppm.

APPENDIX B

RESULTS OF METALS AND ORGANICS ANALYSES ON BIOTA FROM
MITCHELL BRANCH AND POPLAR CREEK

Table B-1. Metals in resident fish from Mitchell Branch, May 6, 1987

Species ^a	WGT ^b	LGTH ^c	ID ^d	SEX ^e	AG	AS	BE	CD	CR	CU	LI	NI	PB	SB	SE	TL	ZN
REDBRE	8.6	8.0	MB2	IM	<0.05	<0.1	<0.05	0.07	<0.1	0.62	<0.5	<1	0.04	<0.2	0.18	<0.2	5.2
REDBRE	3.2	5.6	MB3	IM	<0.05	<0.1	<0.05	0.07	<0.1	0.79	<0.5	<1	0.02	<0.2	0.28	<0.2	4.2
REDBRE	4.6	6.5	MB5	IM	<0.05	<0.1	<0.05	0.09	<0.1	1.20	<0.5	<1	<0.02	<0.2	0.43	<0.2	6.0
REDBRE	4.8	6.4	MB6	IM	<0.05	<0.1	<0.05	0.03	<0.1	<0.30	<0.5	<1	0.04	<0.2	0.82	<0.2	5.7
REDBRE	3.3	5.9	MB9	IM	<0.05	<0.1	<0.05	0.02	<0.1	<0.30	<0.5	<1	0.33	<0.2	0.61	<0.2	5.8
REDBRE	8.6	8.0	MB11	IM	<0.05	<0.1	<0.05	0.03	<0.1	0.88	<0.5	<1	0.03	<0.2	0.65	<0.2	5.8
CKCHUB	8.6	9.4	MB13	IM	<0.05	<0.1	<0.05	0.02	<0.1	0.36	<0.5	<1	0.03	<0.2	0.85	<0.2	12.0

^aREDBRE - Redbreast sunfish (*Lepomis auritus*); CKCHUB - Creek chub
(*Semotilus atromaculatus*).

^bWGT = Weight, gm.

^cLGTH = Total length, cm.

^dID = Fish identification number.

^eIM = Immature.

Table B-2. Mercury (Hg, ppm wet weight) in resident fishes from Mitchell Branch, May 6, 1987

Species ^a	WGT ^b	LGTH ^c	ID ^d	SEX ^e	Hg
REDBRE	5.7	6.8	MB1	IM	0.18
REDBRE	8.6	8.0	MB2	IM	0.16
REDBRE	4.5	6.5	MB4	IM	0.17
REDBRE	4.4	6.5	MB7	IM	0.18
REDBRE	4.4	6.1	MB8	IM	0.18
REDBRE	3.3	5.9	MB10	IM	0.15
REDBRE	2.5	5.3	MB12	IM	0.17
CKCHUB	8.6	9.4	MB13	IM	0.29

^aREDBRE = Redbreast sunfish (*Lepomis auritus*);
 CKCHUB = Creek chub (*Semotilus atromaculatus*).

^bWGT = Weight, gm.

^cLGTH = Total length, cm.

^dID = Fish identification number.

^eIM = Immature.

Table B-3. Mercury (Hg, ppm wet weight) and PCBs (ppm wet weight) in sunfishes from Poplar Creek, June 1, 1987

Site ^a	Species ^b	Sex ^c	ID ^d	WGT ^e	LGTH ^f	Hg	PCB-1254 ^g	PCB-1260 ^h	TOTAL PCB ⁱ
PCK 6.9	BLUGIL	F	7052	93.0	17.6	0.57	0.13	0.19	0.32
PCK 6.9	BLUGIL	F	7054	71.1	15.6	0.57	0.03	0.02	0.05
PCK 6.9	BLUGIL	F	7055	63.5	15.3	0.27	0.18	0.11	0.39
PCK 6.9	BLUGIL	M	7056	85.3	16.7	0.24	0.02	0.07	0.09
PCK 6.9	BLUGIL	F	7057	104.2	17.3	0.65	0.15	0.38	0.53
PCK 6.9	BLUGIL	M	7061	102.1	17.0	0.38	0.04	0.15	0.19
PCK 6.9	BLUGIL	F	7077	75.5	15.9	0.38	0.07	0.07	0.14
PCK 6.9	BLUGIL	F	7078	42.8	13.8	0.33	0.03	0.08	0.11
PCK 6.9	REDBRE	M	7025	141.5	19.5	0.38	0.17	0.29	0.46
PCK 6.9	REDBRE	F	7638	15.2	9.6	--	0.11	0.18	0.29
PCK 6.9	REDBRE	F	9157	41.0	13.3	0.27	0.16	0.18	0.34
PCK 6.9	REDBRE	F	9177	48.4	13.6	0.19	0.06	0.12	0.18

^aPCK 6.9 = Poplar Creek kilometer 6.9.

^bBLUGIL = Bluegill sunfish (*Lepomis macrochirus*) REDBRE = Redbreast sunfish (*L. auritus*).

^cF = Female; M = male.

^dID = Fish identification number.

^eWGT = Weight, gm.

^fLGTH = Total length, cm.

^gPCB quantified against commercial mixture Arochlor-1254 as standard.

^hPCB quantified against commercial mixture Arochlor-1260 as standard.

ⁱTotal PCB = Sum of PCB-1254 and PCB-1260.

Table B-4. Results of caged clam (*Corbicula fluminea*) analyses by capillary column GC/MS and GC/ECD for screening organic priority pollutants. Only the detected compounds are listed

Site ^a	ID ^b	Date	PCB-1254 ^c	PCB-1260 ^d	Di-n-butylphthalate ^e	Bis (2-ethylhexyl)-phthalate ^e
MIK 0.14	MBKU2	3/18-4/15/87	0.96	<0.10	6.1	4.2
MIK 0.14	MBKU3	3/18-4/15/87	1.70	<0.10	<2.0	<2.0
MIK 0.14	MBKL2	3/18-4/15/87	3.0	0.13	<2.0	<2.0
MIK 0.14	MBKL4	3/18-4/15/88	3.5	0.16	<2.0	<2.0
MIK 0.14	MBKL5	3/18-4/15/87	0.35	0.22	9.0	6.4
BVK	BVK-1	3/18/87	<0.10	<0.10	<2.0	<2.0
BVK	BVK-2	3/18/87	<0.10	<0.10	<2.0	<2.0

^aMIK 0.14 = Mitchell Branch kilometer 0.14; BVK = Beaver Creek.

BVK = Beaver Creek.

^cID = Sample identification number.

^ePCB-1254 quantified against commercial mixture Arochlor-1254 as standard, ppm wet weight.

^dPCB-1260 quantified against commercial mixture Arochlor-1260 as standard, ppm wet weight.

^eppm wet weight.

Table B-5. List of organic compounds that were not detected in any of the clams analyzed in GC/MS

Compound	Detection limit (ppm wet wt)
Phenol	<2.0
Bis (2-Chloroethyl)ether	<2.0
2-Chlorophenol	<2.0
1,3-Dichlorobenzene	<2.0
1,4-Dichlorobenzene	<2.0
Benzyl alcohol	<2.0
1,2-Dichlorobenzene	<2.0
2-Methylphenol	<2.0
Bis (2-chloroisopropyl)ether	<2.0
4-Methylphenol	<2.0
N-nitroso-di-n-propylamine	<2.0
Hexachloroethane	<2.0
Nitrobenzene	<2.0
Isophorone	<2.0
2-Nitrophenol	<2.0
2,4-Dimethylphenol	<2.0
Benzoic acid	<10
Bis (2-chloromethoxy)methane	<2.0
2,4-Dichlorophenol	<2.0
1,2,4-Trichlorobenzene	<2.0
Naphthalene	<2.0
4-Chloroaniline	<2.0
Hexachlorobutadiene	<2.0
4-Chloro-3-methylphenol	<2.0
2-Methylnaphthalene	<2.0
Hexachlorocyclapentadiene	<2.0
2,4,6-Trichlorophenol	<2.0
2,4,5-Trichlorophenol	<10
2-Chloronaphthalene	<2.0
2-Nitroaniline	<10
Dimethylphthalate	<2.0
Acenaphthylene	<2.0
2,6-Dinitrotoluene	<2.0
3-Nitroaniline	<10
Acenaphthene	<2.0
2,4-Dinitrophenol	<10
4-Nitrophenol	<10
Dibenzofuran	<2.0
2,4-Dinitrotoluene	<2.0
Diethylphthalate	<2.0

Table B-5. (Continued)

Compound	Detection limit (ppm wet wt)
4-Chlorophenyl-phenylether	<2.0
Fluorene	<2.0
4-Nitroaniline	<10
4,6-Dinitro-2-methylphenol	<10
N-Nitrosodiphenylamine	<2.0
4-Bromophenyl-phenylethene	<2.0
Hexachlorobenzene	<2.0
Pentachlorophenol	<10
Phenanthrene	<2.0
Fluoranthene	<2.0
Pyrene	<2.0
Butylbenzophthalate	<2.0
3,3'-Dichlorobenzidene	<4
Benzo(a)anthracene	<2.0
Chrysene	<2.0
Di-n-octylphthalate	<2.0
Benzo(b)fluoranthene	<2.0
Benzo(k)fluoranthene	<2.0
Benzo(a)pyrene	<2.0
Indeno(1,2,3-cd)pyrene	<2.0
Dibenz(ah)anthracene	<2.0
Benzo(ghi)perylene	<2.0
Alpha-bhc	<0.005
Beta-bhc	<0.005
Delta-bhc	<0.005
Gamma-bhc	<0.005
Heptachlor	<0.005
Aldrin	<0.005
Heptachlor epoxide	<0.005
Endosulfan-i	<0.005
Dieldrin	<0.010
4,4'-dde	<0.010
Endrin	<0.010
Endosulfan-ii	<0.010
4,4'-ddd	<0.010
Endosulfan sulfate	<0.010
4,4'-ddt	<0.010
Methoxychlor	<0.050
Endrin ketone	<0.010
Alpha-chlordane	<0.050
Gamma-chlordane	<0.050
Toxaphene	<0.100

Table B-5. (Continued)

Compound	Detection limit (ppm wet wt)
PCB-1016	<0.050
PCB-1221	<0.050
PCB-1232	<0.050
PCB-1242	<0.050
PCB-1248	<0.050

Table B-6. Results of PCB (ppm wet weight) analyses of caged clams (*Corbicula fluminea*) by packed column GC/ECD, March-April and July-August, 1987

Site ^a	ID ^b	Date	PCB-1254 ^c	PCB-1260 ^d	TOTAL PCB ^e
MIK 0.14	MBKU1	3/18 - 4/15/87	4.6	<0.05	4.6
MIK 0.14	MBKU4	3/18 - 4/15/87	3.9	<0.05	3.9
MIK 0.14	MBKL1	3/18 - 4/15/87	3.0	<0.05	3.0
MIK 0.14	MBKL3	3/18 - 4/15/87	3.4	<0.05	3.4
MIK 0.14	MBKL6	3/18 - 4/15/87	4.6	<0.05	4.6
BVK	BVK-3	3/18/87	0.28	<0.01	0.28
BVK/1504	1504-3	3/18 - 4/15/87	0.16	<0.01	0.16
MIK 0.14	K25-1	7/15 - 8/17/87	2.4	<0.01	2.4
MIK 0.14	K25-2	7/15 - 8/17/87	2.7	<0.01	2.7
MIK 0.14	K25-1	7/15 - 8/17/87	2.5	<0.01	2.5
BULLRN	BR-1	7/15/87	0.05	0.01	0.06
BULLRN	BR-2	7/15/87	0.02	0.02	0.04

^aMIK 0.14 - Mitchell Branch kilometer 0.14; BVK - Beaver Creek; BVK/1504 - Beaver Creek clams held at Oak Ridge National Laboratory in Building 1504 source water; BULLRN - Bull Run.

^bID - Sample identification number.

^cPCB-1254 quantified against commercial mixture Arochlor-1254 as standard.

^dPCB-1260 quantified against commercial mixture Arochlor-1260 as standard.

^eSum of PCB-1254 and PCB-1260.

APPENDIX C

ANNUAL PRODUCTION/BIOMASS RATIOS AND COHORT PRODUCTION
INTERVALS FOR BENTHIC MACROINVERTEBRATES

Table C-1. Annual production/biomass ratios (P/B) and cohort production intervals (CPI) used for annual production estimates of Mitchell Branch benthic macroinvertebrates, August 1986 through July 1987. Unless otherwise indicated, values given for the higher taxa were also used for their respective lower taxa

Taxon	P/B ^a	CPI (mo) ^b
Turbellaria	6.0	10.0
Nematoda	10.0	6.0
Oligochaeta	10.0	6.0
Isopoda	5.0	12.0
Amphipoda	5.0	12.0
Decapoda	1.5	36.0
Hydracarina	10.0	6.0
Insecta		
Collembola	10.0	6.0
Ephemeroptera		
Baetidae	10.0	6.0
Baetis ^c	10.5	3.0
Caenidae	10.0	6.0
Ephemerellidae	6.0	10.0
Ephemeridae	5.0	12.0
Leptophlebiidae	6.5	9.0
Odonata	5.0	12.0
Aeshnidae	2.5	24.0
Calopterygidae	5.0	12.0
Coenagrionidae	5.0	12.0
Cordulegastridae	2.5	24.0
Gomphidae	2.5	24.0
Libellulidae	2.5	24.0
Plecoptera	5.0	12.0
Capniidae	10.0	6.0
Chloroperlidae	5.0	12.0
Leuctridae	6.5	9.0
Nemouridae	10.0	6.0
Perlidae	5.0	12.0
<i>Eccopectura xanthenes</i>	2.5	24.0
Hemiptera		
Gerridae	40.0	1.5
Veliidae	40.0	1.5
Megaloptera		
<i>Nigronia fasciatus</i>	2.5	24.0

Table C-1. (Continued)

Taxon	P/B	CPI (mo)
Megaloptera (cont.)		
<i>Sialis</i>	5.0	12.0
Trichoptera	5.0	12.0
Hydropsychidae		
<i>Cheumatopsyche</i>	10.0	6.0
<i>Diplectrona modesta</i>	5.0	12.0
Leptoceridae	7.5	8.0
Limnephilidae		
<i>Goera</i>	6.0	10.0
<i>Hydatophylax</i>	5.0	12.0
<i>Neophylax</i>	6.0	10.0
<i>Pycnopsyche</i>	5.0	12.0
Molannidae	6.0	10.0
Philopotamidae	10.0	6.0
Psychomyiidae	5.0	12.0
Lepidoptera	6.0	10.0
Coleoptera		
Dryopidae	2.5	24.0
Dytiscidae	10.0	6.0
Elmidae		
<i>Dubiraphia</i>	2.5	24.0
<i>Optioservus</i>	2.5	24.0
<i>Stenelmis</i> ^C	1.8	24.0
Hydrophilidae	21.0	2.8
Ptilodactylidae	2.5	24.0
Staphylinidae	5.0	12.0
Diptera	5.0	12.0
Ceratopogonidae	10.0	6.0
Chaoboridae	10.0	6.0
Chironomidae		
Chironominae		
Chironomini	15.0	4.0
Tanytarsini	26.0	2.3
Diamesinae	10.0	6.0
Orthocladinae	24.0	3.0
Tanypodinae	17.1	3.5
Dolichopodidae	5.0	12.0
Dixidae	5.0	12.0
Empididae	5.0	12.0
Muscidae	5.0	12.0
Simuliidae	15.0	4.0
Stratiomyidae	5.0	12.0
Tabanidae	6.5	9.0
Tipulidae	5.0	12.0

Table C-1. (Continued)

Taxon	P/B	CPI
Gastropoda		
<i>Ferrissia</i>	8.6	7.0
<i>Fossaria</i>	3.3	18.0
<i>Physella</i>	3.3	18.0
Pelecypoda	3.3	18.0

^aValues derived from a theoretical annual P/B of 5.0 and corrected for CPI (Waters 1977; Waters 1979).

^bValues derived from published life history information.

^cEmpirically derived from other BMAP studies on the DOE Oak Ridge Reservation (J. G. Smith, unpublished data).